RESOURCE RECOVERY FROM SANITATION TO AMPLIFY DEVELOPMENT: NAVIGATING GLOBAL AND LOCAL POSSIBILITIES

BY

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DISSERTATION

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ABSTRACT

Recovery of human-derived resources (e.g., nutrients, energy) from sanitation systems has emerged as an approach that may generate progress toward multiple Sustainable Development Goals (SDGs). However, persistent uncertainties and concerns (e.g., economics, social appropriateness) across a variety of scales and settings limit implementation, and failures are common. Decision-makers require more rigorous methods and tools that generate evidence to characterize various sanitation, recovery, and reuse options. Therefore, the overarching goals of this work are to explore and quantify the possibilities, benefits, and challenges associated with resource recovery, and to establish quantitative models and conceptual frameworks capable of contributing to global and local paths forward for sanitation. Specific objectives include (i) estimating recovery's impacts on resource access; (ii) developing methods to define spatial co-location and transport requirements; (iii) employing spatial methods to assess soil suitability of recovery products; (iv) developing a conceptual framework linking recovery with ecosystem services; and (v) developing a social-ecological systems framework that defines sanitation as a human-derived resource system and supports multidimensional analysis across contexts.

The first three objectives examine particular topics associated with resource recovery at a global scale by developing and implementing quantitative models. These analyses point toward locality-specific strategies for deriving the greatest benefit from sanitation investments, while also identifying overarching trends to guide international research efforts. First, resource recovery from sanitation systems that will need to be installed to achieve sanitation SDG targets may considerably improve access to agricultural nutrients and household energy in low-income countries, six of which could double or offset all projected nutrient and electricity use. Global potential nutrient gains are an order of magnitude larger than those for electricity. Second, closing urban nutrient cycles will require transport of nutrients from cities to surrounding cropland. Estimated transport distances across 56 of the world's largest cities span two orders of magnitude

and are often shorter among European, African, and Asian cities due to factors such as high local cropland density and nutrient-intensive crops. The energy requirements associated with transporting nutrients may constrain whether certain recovery strategies and products (e.g., reclaimed wastewater, sludge) are locally feasible. Finally, the agronomic value of nutrient application depends upon interactions between product chemistry and soil context. For example, alkaline products (e.g., struvite) may be particularly beneficial when applied to acidic soils in a country like Uganda but potentially detrimental in other contexts like the southwest United States.

The final two objectives reflect the need to develop broadly applicable conceptual frameworks that enumerate possibilities and support holistic and contextual assessment of sanitation options. The first framework characterizes links between resource recovery from sanitation and ecosystem services to shed light on the viability of exploring synergistic interactions between engineered and natural systems. Bridging these fields may create opportunities to support goals related to climate regulation, soil conservation, and water quality, among others. A spatial analysis further demonstrates resource recovery's potential to contribute to different regional ecosystems across the globe. The second framework envisions sanitation as a distinct social-ecological system type centered on human-derived resources (e.g., nutrients, energy), functioning within a broader social, economic, and environmental setting. A set of key variables at different scales are identified and placed in the framework structure, which is applied to assess alternative sanitation scenarios in a specific context (Bwaise, Uganda) to reveal multi-dimensional tradeoffs and the impacts of individual processes on system outcomes.

Overall, this work suggests that resource recovery from sanitation can create numerous opportunities to advance sustainable development goals, but a number of local and global issues affect the feasibility and potential impact of various recovery and reuse strategies. Moving forward, integrating the analyses and frameworks developed here and implementing them across various settings can support understanding and decision-making around sanitation and resource recovery to design effective systems that ensure a more sustainable future.

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TABLE OF CONTENTS

CHAPTER 1: INTRODUCTION AND BACKGROUND

Summary of Motivation and Objectives

Historically, sanitation systems were developed primarily to mitigate environmental and human health risks, but long-established conventional systems do not fully address other dimensions of sustainability. A tradition of sewage and activated sludge treatment systems (sometimes including nutrient removal to address the effects of nitrogen and phosphorus discharge) characterizes sanitation in industrialized countries[1–5]. Alternatively, in settings where sanitation coverage remains low and where limited access to water, energy, and other resources makes conventional wastewater treatment less feasible[6–8], simple, low-cost, onsite sequestration systems (e.g., pit latrines) are commonly implemented to meet sanitation targets set forth by the United Nations through the Sustainable Development Goals (SDGs)[9–12]. The inertia caused by the continued use of these approaches has restricted the application of processes better equipped to capitalize on the resources (e.g., nutrients, organic matter) present in human excreta and create positive interactions between sanitation and other dimensions of sustainable development (e.g., agriculture, nutritional security, energy access, climate change mitigation)[7,13–28].

Consequently, resource recovery has emerged as an important consideration in recent sanitation research[7,13–17,28], but uncertainties and concerns across a variety of scales and settings limit its implementation. While global assessments of nitrogen and phosphorus in human excreta exist[4,29–34], other resources (potassium, energy) are less well-characterized. Additionally, even existing estimates do not typically provide information at a scale relevant to individual countries or communities, and they often neglect elements such as data variability, recovery efficiencies, soil context, and spatial connectivity between recovered resources and locations where they are needed. The question of spatial co-location of nutrients with agricultural needs is especially relevant for cities[4,34–36], but the feasibility of recycling resources (e.g., economic viability, transport logistics) in these contexts remains unknown. Simultaneously, particularly in resource-limited settings, implementation of resource recovery often faces challenges of appropriateness (technical, economic, environmental, or social) that limit possible benefits. Numerous factors can contribute to system failures[8,27,37] and are particularly common when actors unfamiliar with the local setting promote and apply inappropriate technologies or approaches[8,11,27]. To combat these drivers of failure, decision-makers require more rigorous information and tools to present a variety of sanitation, recovery, and reuse options and evaluate their performance across relevant dimensions of sustainability[19,27,38]. A wide array of methods (e.g., life cycle assessment, multi-objective optimization, material flow analysis) could be (and in some cases have been) applied to sanitation planning[39–46]. However, those applications have not resulted in the development of generalized frameworks that can support decision-making around sanitation and sustainability in resource-limited settings[39,47].

Therefore, the overarching goals of this work were to explore and quantify the possibilities, benefits, and challenges associated with Resource Recovery from Sanitation (RRS), and to establish quantitative models and conceptual frameworks capable of contributing to global and local paths forward for sanitation. At its core, this work addresses a critical barrier to the widespread development of resource recovery technologies as sustainable engines of economic, environmental, and social progress: a lack of robust analysis to identify and navigate tradeoffs surrounding RRS (particularly in resource-limited settings), leading to a prevalence of mismatches in which inappropriate systems are implemented. This barrier stems from inertia and over-activism: inertia to continue using resource-draining sanitation solutions that do not address water stress, energy efficiency, and environmental sustainability concerns; and over-activism that promotes “one size fits all” technologies while lacking adequate evidence for widespread application. Accordingly, this barrier has contributed to a limited understanding of whether RRS is feasible and able to generate significant positive impacts across various settings and scales.

The specific objectives of this work were: (i) to estimate potential impacts of RRS on access to resources at global, regional, and national levels; (ii) to develop and implement methods to define spatial characteristics and transport requirements of RRS; (iii) to further develop and employ spatial methods to assess the suitability of RRS processes relative to local soil context; (iv) to develop a conceptual framework to characterize mutually-beneficial interactions between RRS and ecosystem services; and (v) to develop a generalized social-ecological systems framework for RRS that can be applied in contextual settings to explore the multidimensional impacts of sanitation possibilities and evaluate their sustainability. The research objectives were accomplished by leveraging global datasets within quantitative modeling frameworks to explore the potential impacts and spatial complexities of RRS; by integrating these analyses with reviews of relevant literature to develop conceptual frameworks that define potential interactions across sanitation, ecological, and social systems; and by integrating sanitation system and resource flow modeling with field research in an informal settlement in Kampala, Uganda to inform decision-making by local stakeholders.

The Resources Available in Human Excreta

Substantial resource quantities are embedded in human excreta and could be recovered. The chemical compositions of human urine and feces have been characterized in various settings, although most data come from Europe and North America[48]. Nutrient and energy excretion depend on dietary intake[20,29,48], and highly variable quantities of excreted nitrogen, phosphorus, potassium, and chemical energy have been reported[17,20,48,49]. Generally, most excreted energy is contained in feces as organic matter[20,48], but fecal excretion accounts for only 2-10% of the total caloric energy that is consumed (most is stored or oxidized to drive biological processes)[50]. In contrast, adult nutrient ingestion and excretion are essentially in balance[51,52], with urine containing most nitrogen and potassium and approximately half of phosphorus[20,48]. To go beyond global averages, nutrient and energy excretion can be estimated from country-level protein and caloric intake data[53] using conversion factors[17,20,29,50,54–58]. Resources can then be recovered using various strategies[16,59–61], although potassium recovery options remain limited[62]. Water recovery could also occur, but large gaps surround information on global, regional, and national wastewater production and treatment, with only 30% of 181 assessed countries having complete data[63].

Connecting Resource Recovery with Global Needs through Quantitative Modeling

The growing global population requires ever-greater food and energy supplies, which must be provided by resource systems already straining planetary boundaries[3,64]. Global flows of nitrogen and phosphorus (key fertilizer nutrients) currently exceed defined thresholds, outside of which dramatic ecosystem changes could result (e.g., excessive nutrient loads in aquatic ecosystems can cause eutrophication and create dead zones)[3]. Fertilizer production now represents 1% of total global energy demand[65], mostly due to the conversion of atmospheric nitrogen gas to ammonia through the Haber-Bosch process[26,65,66]. Moreover, phosphorus and potassium supplies are non-renewable and regionally concentrated (and therefore dependent on geopolitics)[30,67]. Global energy demand is met predominantly through fossil fuel use[68], while solid biomass (e.g., firewood) may account for 80-90% of total energy use in the poorest countries in Africa, Asia, and Latin America[68].

As a consequence of these imbalanced flows and production patterns, the SDGs call for improved agriculture (SDG 2) and energy (SDG 7) systems, both to increase resource access in resource-limited settings and replace unsustainable use in other areas[12]. Simultaneously, SDG Targets 6.2 and 6.3 endeavor to achieve universal sanitation coverage by 2030 and halve the proportion of untreated wastewater[12,69] – targets facing global shortfalls of 2.3 billion people without at least basic sanitation[69,70] and an additional 1.5 billion without wastewater treatment[71]. Therefore, using resource recovery systems to meet sanitation targets may improve progress toward multiple SDGs (e.g., the total phosphorus in human excreta in 2009 represented approximately 22% of worldwide phosphorus demand[29]). A global analysis of resource recovery impacts associated with the SDG sanitation targets (Obj. i) will provide a quantitative foundation for critical work surrounding appropriate implementation and local decision-making.

As global urbanization trends concentrate more people in densely-populated areas, examining spatial links between resource recovery and reuse will be critical. More than half of the global population now lives in urban settings, which will continue growing in the coming decades (especially in Asia and Africa)[72]. Cities have historically relied on surrounding rural areas for services such as food production[72], although urban agriculture has recently received attention as a mechanism for improving food security, especially among low-income urban residents[73–75]. However, urban agriculture is unlikely to significantly improve food supplies due to space limitations, and these constraints are particularly restrictive in poorer countries[74,75]. Accordingly, recovered urban nutrients may need to travel considerable distances. Work that characterizes the degree of co-location between areas of recovery and potential reuse to estimate required transport distances is critical to evaluating the viability of circular economies that utilize nutrients recovered from sanitation. For example, in the United States, 74% of corn's phosphorus demand could be met by recovering human, animal, and food waste in the same county, while surplus phosphorus could then be transported an average of 302 kilometers to meet demand in the corn belt[76]. Analyzing resource co-location for a variety of cities around the world (Obj. ii) will provide valuable information surrounding the constraints associated with recovering and recycling nutrients in various forms (e.g., relatively bulky digested sludge or more concentrated, nutrient-dense crystalline products such as struvite).

When applied to cropland, different types of nutrient products (e.g., reclaimed wastewater, digested sludge, compost, source-separated urine, crystalline products) will behave differently from one another and may have divergent impacts on crop production, nutrient use efficiency, and soil quality[77–83]. However, little work has been done on a global level to assess these potential interactions between product chemistry and soil context to assess local suitability. Thus, soil context could play an important role in driving decisions around whether nutrient recovery should be pursued and what recovery products should be generated and/or reused in a given locality. Developing a globally-relevant methodology for analyzing this issue and applying it for a variety of nutrient recovery products (Obj. iii) will provide a foundation for characterizing the value proposition of nutrient recovery with respect to contextual soil conditions.

Connecting Resource Recovery with Local Needs through Conceptual Frameworks

High-level assessments of RRS provide foundational support for site-specific research and implementation, where conceptual frameworks that enumerate possibilities and holistically assess options while accounting for contextual drivers are needed. In the past, sanitation provision in resource-limited communities typically occurred through approaches providing "one size fits all" systems ignoring local context[11,84], often leading to failure[11,37,84]. Therefore, approaches are needed that present a wide variety of options for sanitation and resource recovery, acknowledge the complex interactions across sanitation and other context-specific systems, quantitatively assess the locality-specific characteristics of alternatives, and engage stakeholders to understand local factors[37,41,49,85,86].

In addition to specific opportunities around improved water quality, agricultural production, and energy access, recovered resources may also contribute to mutually-beneficial interactions with a number of ecosystem services (ES)[87]. ES such as food and water provisioning, nutrient cycling, and climate regulation can contribute to several SDGs, including those related to reducing hunger, sustaining aquatic and terrestrial life, ensuring clean water, developing sustainable cities, and promoting climate action[88,89]. Recovered resources represent materials society can contribute back to ecosystems, thereby supporting a positive cycle of reciprocal benefits (e.g., by enhancing services such as erosion control and food provisioning through organic matter and nutrient application). Indeed, integrated design paradigms can expand the engineering design space to harmonize technological and natural processes and develop synergistic approaches to meet societal and ecosystem needs[90,91]. However, sanitation or water management frameworks explicitly incorporating ES beyond water quality improvement[92] remain rare. Frameworks that integrate the value of enhancing ES (Obj. iv) can represent a more holistic view of sanitation –

one that may reveal greater opportunities for resource recovery in a variety of contexts, but particularly in settings with considerable ecological assets but limited economic means.

Finally, holistically understanding and evaluating sanitation's role within a given context requires a more comprehensive framework that views a sanitation system as a sequence of integrated but distinct functional units (the user interface; onsite collection/storage/treatment; conveyance; centralized treatment; reuse/disposal)[49,84,85] and also embeds sanitation within a broader social-ecological system (SES)[93–95]. Envisioning sanitation as an integrated system of sub-processes allows for holistic consideration of one component's effects on other parts of the system, providing insight into design and evaluation that may not be apparent from assessments of individual components[96]. Beyond the sanitation system itself, SES frameworks seek to understand a complex whole through knowledge of specific variables and relationships, and they are typically used for common-pool resources (e.g., trees, fish stocks) requiring collective management[95]. A framework focused on sanitation presents a distinct case, in which each person contributes to flows of resources (e.g., nutrients, energy) that can be captured and converted into productive forms through RRS. Sanitation systems become integrated with agriculture and energy systems, a concept only beginning to be studied among households in sub-Saharan Africa[46]. Developing a SES framework for sanitation and applying it to study the multidimensional impacts of various sanitation possibilities within a specific context (Obj. v) will integrate an understanding of the opportunities and challenges evaluated under previous objectives with stakeholder engagement and locality-specific quantitative modeling to evaluate resource recovery, economic, and environmental outcomes. This work can support local decision-makers in their efforts to learn about relevant system dynamics, explore options, understand the consequences of those options, and make informed decisions[97].

Organization of this Dissertation around the Research Objectives

This dissertation presents work related to the five research objectives, exploring the possibilities and challenges associated with RRS across global and local scales (Figure 1.1). First, multiple global-level analyses are developed and implemented to characterize large-scale trends around the potential impacts and viability of different processes and products associated with RRS. Quantitatively estimating the possible impacts of RRS on the sustainable development of nations and regions (Obj. i) provides an initial understanding of how resource recovery may amplify progress toward development goals, especially in resource-limited settings (Chapter 2). Given the potential for nutrient recovery in particular to have substantial impacts on global resource access, the two subsequent chapters focus on key issues surrounding the spatial feasibility (Obj. ii) and soil suitability (Obj. iii) of nutrient recovery and reuse in agriculture. Cities represent large concentrations of nutrients recoverable from sanitation, but the spatial patterns associated with transport to rural agricultural land may constrain the types of recovery processes and products that are viable (Chapter 3). When nutrient products are applied to cropland, their chemistry may interact with soil conditions in beneficial or detrimental ways, potentially impacting agricultural productivity (Chapter 4). Generally, these global analyses may support international pathways toward improved resource sustainability and provide quantitative information to augment local understanding around the opportunities and challenges associated with RRS.

The global analyses also begin to reveal some of the mechanisms through which RRS may interact with local ecological, economic, and social systems to affect multiple dimensions of sustainability. Through literature review and contextual experience, two generalized conceptual frameworks are developed to explore these multi-faceted relationships. Mutually beneficial interactions between RRS and ecosystem services may offer opportunities to enhance the societal value of ecological resources while reducing unintended negative environmental consequences sometimes associated with sanitation (Chapter 5). The second framework takes a broader view of sanitation as a social-ecological system (SES), where sanitation and resource recovery systems interact with and are embedded in complex economic, environmental, social, and political structures. This SES framework is developed and then applied in an informal settlement in Kampala, Uganda by integrating field data collection and stakeholder engagement with quantitative modeling of resource flows, economics, and environmental impacts (Chapter 6). These frameworks can help to support decision-making by various sub-national stakeholders (e.g., groups within individual communities, city officials, regional planners) and inform a global understanding of how specific context can influence the outcomes associated with sanitation and resource recovery[8,19,41,84,98–102]. A final chapter summarizes key findings from all the studies and considers the significance of this line of research moving forward (Chapter 7).

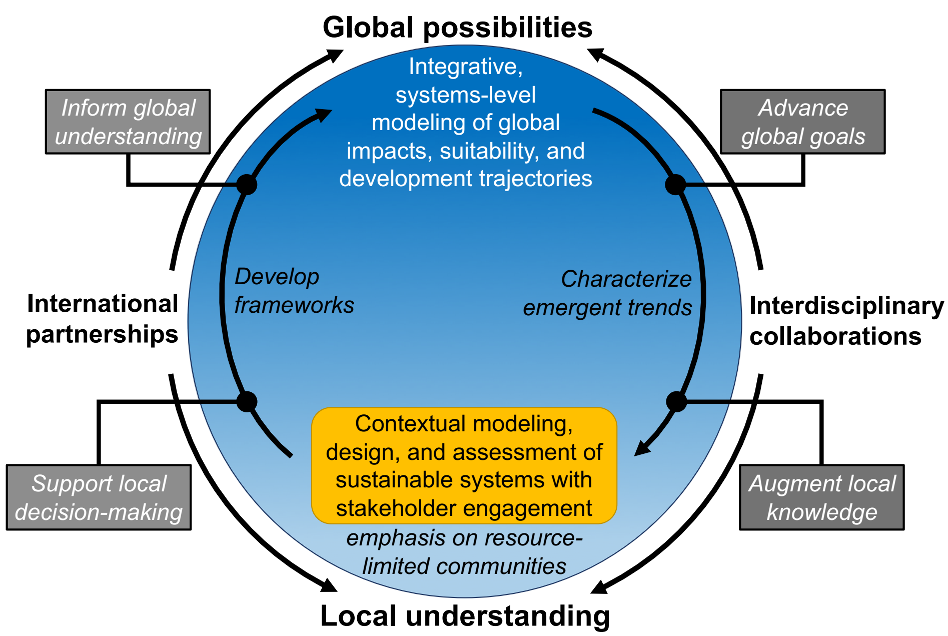

Figure 1.1. A conceptual figure illustrating how analyses around RRS at global and local scales can interact and feed into one another. Study across multiple scales and settings is needed to fully explore the multidimensional nature of sanitation.

CHAPTER 2: AMPLIFYING PROGRESS TOWARD MULTIPLE DEVELOPMENT GOALS THROUGH RESOURCE RECOVERY FROM SANITATION[a]

Introduction

The United Nations' Sustainable Development Goals (SDGs) recognize the need to eliminate current sanitation gaps to mitigate environmental and human health risks. SDG Targets 6.2 and 6.3 endeavor to achieve universal sanitation coverage and halve the proportion of untreated wastewater by 2030[12], respectively, lofty goals considering current global shortfalls of 2.3 billion people without basic sanitation access[103] and an additional 1.5 billion served by sewers without wastewater treatment[71]. Currently, the largest sanitation gaps exist in the least-developed countries (LDCs, as defined in the 2017 SDG Report)[104], where inadequate availability of fertilizers, household electricity, and non-solid cooking fuels[53,105,106] (Figure S3) reflects limited progress toward sustainable agriculture (SDG 2) and energy access (SDG 7).

Concurrently, the growing global population requires ever-larger quantities of food, energy, and other resources, which must be met by agricultural, economic, and infrastructure systems already straining planetary boundaries[3,64]. Global flows of nitrogen and phosphorus (key fertilizer components) currently exceed defined thresholds, outside of which dramatic ecosystem changes could result[3]. Additionally, converting atmospheric nitrogen gas to ammonia fertilizer through the Haber-Bosch process is energy-intensive[66], and phosphate rock is a non-renewable resource concentrated predominantly in a few countries, linking the mineral's availability with international politics[30]. Consequently, alternative and renewable nutrient and energy sources are required, especially in resource-limited settings. Given annual quantities of nitrogen (1.6-7.4 kg), phosphorus (0.4-1.0 kg), potassium (0.4-2.3 kg), and chemical energy in reduced carbon (11-123

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kWh) present in one person's excreta (with variations dependent upon diet)[17,20], resource recovery from sanitation presents a possible platform for amplifying countries' efforts to meet multiple SDGs (Table S1). Access to renewable sources of nutrients and energy could enhance the global resource systems sustaining human development by increasing use in resource-limited settings, and replacing extracted resources in highly-industrialized locations.

Global estimates of reuse from human sanitation are highly variable but suggest 0-15% of nitrogen and 0-55% of phosphorus are currently recycled to cropland[4,30,31,34,53,107–109] (to our knowledge, global potassium recovery has not been assessed). Phosphorus recovery is likely greater because sewage sludge, high in phosphorus but low in nitrogen and potassium[110], is currently a common source of recycled nutrients. Simultaneously, large quantities of nutrients and energy are lost to the environment due to inadequate treatment, especially in low- and middle-income countries[111]. Looking forward, sanitation's potential to contribute to improved resource accessibility and sustainability is being investigated at local and global scales. For example, the total phosphorus available globally in human excreta in 2009 represented approximately 22% of worldwide phosphorus demand[29]. Improving sustainability of urban sanitation options has received particular attention[49,112], and research initiatives surrounding urban sanitation demonstrate context-specific efforts to recycle human excreta using innovative management models (e.g., container-based sanitation in Haiti[113,114]). The recently established SDGs present a timely opportunity to investigate the multidimensional effects of meeting sanitation targets while integrating resource recovery.

Accordingly, we estimated potential impacts of achieving SDG Targets 6.2 and 6.3 by 2030 using resource recovery sanitation systems, specifically on nutrient and household electricity use in countries across the economic spectrum. Ultimately, we identify countries and regions where nutrient and/or energy recovery can generate meaningful amplifying effects related to SDGs 2 (sustainable agriculture) and 7 (energy access), providing guidance for local, national, and international actors to improve sanitation while simultaneously strengthening agriculture and energy systems. To assess possible benefits of achieving each sanitation-related SDG target, potential nutrient and energy recoveries were estimated from populations projected to be served by three categories of sanitation systems: (i) newly-installed sanitation systems needed to achieve universal sanitation coverage (Target 6.2), (ii) newly-treated wastewater systems needed to halve the proportion of untreated wastewater (Target 6.3), and (iii) existing systems that could be replaced or retrofitted to achieve resource recovery but are not directly linked with the SDGs (Figs. S1-S2). As regulations or recommendations specifying blanket application of particular technologies can hamper development and restrict innovation,[115] we estimate resource recovery potential from each sanitation category on a per capita basis independent of technology. The robustness of the primary scenario's results (based on UN and FAO forecasts) were evaluated using sensitivity and uncertainty analyses (which accounted for recovery efficiency variations across diverse sanitation technologies), and geographical disparities between recoverable nutrients and agricultural needs were considered through a spatial co-location analysis.

Materials and Methods

The modeling methodology used to estimate potential impacts, assuming the SDG sanitation targets are achieved using resource recovery systems, is described below. Briefly, future conditions were estimated using projections of various global datasets, and populations served by each sanitation category were estimated. Quantities of recoverable resources (nitrogen, phosphorus, potassium, energy) were estimated based on country-level dietary intake on a per capita basis (independent of specific recovery technologies and degrees of centralization), scaled according to estimated populations served by each sanitation category, and compared with projected resource use. Uncertainty, sensitivity, and co-location analyses were employed to address issues such as parameter uncertainty (e.g., recovery efficiencies across diverse technologies; Sections S4 and S8), projection robustness, and geographical disparities. Details are provided below, with additional information in the Supporting Materials (Sections S1-S9, Figures S1-S2, Tables S2-S6).

Data Collection. Datasets related to country-level improved sanitation coverage (improved sanitation includes treated and untreated sewer connections, as well as decentralized systems meeting certain standards)[103], sewer connections and wastewater treatment[71], population and urbanization[116], fertilizer consumption and imports[53], food supply[53], household electricity use[106] and access to non-solid fuel[105], technological readiness[117], and spatially-explicit harvested crop area[118] and population distributions[119,120] were collected for the 225 countries and territories included in the sanitation coverage dataset (hereafter referred to collectively as "countries") from relevant literature and international agency databases (the source and quality of each dataset are provided in Table S2). In many cases, temporal ranges of available data did not align (e.g., the most recent sewer connection data is reported for 2010, while improved sanitation coverage is reported up to 2015), and datasets had varying levels of quality (e.g., 2% of all countries lacked sufficient cropland area data for projection and analysis, while over 20% lacked sufficient food supply data; Table S2). For consistency across datasets, only data values from 1990 to 2010 were used in the following procedures. As a preliminary step, the portion of a country's total population served by sewer connections without wastewater treatment was calculated from sewer connection and wastewater treatment data, using a procedure similar to that used by Baum et al.[71] To facilitate comparisons between countries, fertilizer consumption and imports were normalized with respect to cropland area (sum of arable land and permanent crop area), while household electricity use was normalized to population (Section S1).

Projection Scenarios. To develop projections of future conditions in 2030, eight potential scenarios were evaluated (Table S4: one primary scenario and seven alternates to assess the sensitivity of results to alternate projections). In the primary scenario, population and urbanization levels were projected according to the United Nations (UN) 2014 World Urbanization Prospects[116], and, where possible, future fertilizer use, cropland area, and food supply were estimated using regional projections from the Food and Agricultural Organization (FAO)[121]. For countries not explicitly included in given regions, historical data from 1990 to 2010 were linearly extrapolated to 2030, with any negative slopes being replaced with horizontal slopes for conservativeness. This extrapolation method was also used for fertilizer imports and household electricity use, as published projections were not available. To assess the sensitivity of results to different projection assumptions, seven alternate scenarios were developed to encompass a range of possible futures, some of which are highly unlikely and considerably different from the primary scenario (Table S4).

To estimate the implications of achieving the SDG sanitation targets across all eight scenarios, each country's sanitation coverage was assumed to increase from its reported level in 2010 to 100% in 2030 (SDG Target 6.2) and the percentage of each country's 2010 population connected to sewers without wastewater treatment was halved in 2030 (SDG Target 6.3). A starting point of 2010 (the most recent year for which sewer connection and wastewater treatment data were reported) was chosen so that results for both sanitation targets could be directly compared. Additionally, it should be noted that SDG Target 6.2 has transitioned from a technological classification of sanitation systems (previously used to define "improved" systems) to a functional classification (incorporating progression up the sanitation ladder), in which improved, single-household sanitation systems that safely treat, reuse, transport, and/or dispose of excreta (including resource recovery systems) are classified as "safely managed"[115,122].

Results from the primary projection scenario and all alternate scenarios were found to follow similar patterns, with minor exceptions in extreme and unrealistic cases (for example, in Alternate Scenario E, where total resource use is held constant while population increases), showing results to be robust with respect to alternative projection characteristics (Figure S4). Results presented below are from this primary projection scenario (Section S2).

Populations Served by Sanitation Improvements. Potential resource impacts from each sanitation category (newly-installed systems, newly-treated wastewater systems, replaced existing systems) depend upon the number of people using those systems in 2030. First, if a given country's final population in 2030 is greater than its initial population having access to basic sanitation in 2010, the difference between the final and initial populations with sanitation access represents the gap filled by newly-installed systems.

The difference between the initial and final populations served by sewers without wastewater treatment provides an estimate of the population served by newly-treated wastewater systems in 2030, assuming only systems currently discharging untreated wastewater can be converted into newly-treated wastewater systems. New sewer connections for anyone without previous sanitation access would be counted as newly-installed systems, while any transitions from non-sewered sanitation systems to sewer connections would be considered a replacement of existing systems.

Finally, existing systems in 2010 are assumed to be replaced at a constant rate depending on design life. The inverse of design life provides the annual replacement rate, which can be multiplied by the time period (20 years) and the initial population with sanitation access (less those using sewers without wastewater treatment) to estimate the population served by replaced existing systems in 2030. Otherwise, if a country's final population is less than the initial population with sanitation access, the population served by newly-installed systems is zero, and existing systems and/or sewers without wastewater treatment must be abandoned. The following order of abandonment (first to last) was assumed: (i) newly-treated wastewater systems; (ii) existing systems needing replacement; (iii) existing systems not yet requiring replacement; (iv) remaining sewers without wastewater treatment. With this order, countries minimize short-term costs, first abandoning systems otherwise requiring treatment or replacement. Further adjustments may also be needed to account for in-country migration causing different rates of urban and rural population change (Section S3).

Resource Recovery. Resource recovery potential is calculated on a per capita basis and then differentiated based on the projected populations served by each sanitation category. This analysis considered possible recovery of nutrients and energy. Water recovery was not included due to significant data gaps surrounding quantities of wastewater generated and treated[63], and because the amount of water used to manage human excreta can vary tremendously by more than two orders of magnitude (e.g., minimal water is required in decentralized waterless toilets, while >184 liters per day may be used in centralized, sewered systems[123]). Potential per capita energy recovery is dependent first upon caloric intake, estimated using projected caloric supply in 2030 and household consumption waste (the food supply dataset accounts for losses prior to household level). A fraction of caloric intake is excreted as chemical energy in reduced carbon[17,20,50], and a portion of excreted energy can be recovered using one of several technology options[16] (assumed values for energy excretion and recovery can be found in Table S5).

Similarly, nutrient intake was estimated from projected protein and caloric intake: nitrogen intake depends on total protein[57,58]; phosphorus intake depends (separately, as in Mihelcic et al.[29]) on animal and vegetal protein[29,56,58]; and potassium depends on caloric intake[54,55]. In adults, essentially all nitrogen and phosphorus is excreted in urine and feces[51,52], while some potassium is lost through sweat[54,55]. Individual recovery efficiencies were applied for each nutrient (Table S5), representing diverse technology options (e.g., ion exchange, solid precipitation, direct reuse of collected urine and/or feces). As potassium recovery options are currently limited[62], its recovery efficiency considered only direct reuse of source-separated urine.

A country's potential recovery from each sanitation category was calculated as the product of each per capita recovery value and the population served by each category, which was then normalized with respect to projected cropland area (for nutrients) or population (for energy). The potential impact from each sanitation category, signifying either an increase in projected resource use or a replacement of conventional nutrient and/or energy sources (depending on local decisions and resource availability), is calculated by dividing potential recovery by the corresponding fertilizer or household electricity use projected for 2030. Although recovered energy may often be more appropriately used in another form (e.g., methane for cooking, heat to provide treatment within the sanitation system[124]) and household electricity represents a small fraction of total energy use[106,125,126], a lack of data prevented an alternative energy benchmark (Section S4, Table S6).

A weighted average was used to calculate aggregate (N+P+K) nutrient impacts (i.e., the percent impact for each nutrient was multiplied by the mass of projected fertilizer use, the sum across all three nutrients was calculated, and the sum was divided by total projected fertilizer use of all three nutrients; Eq. S12). Possible replacement of imported fertilizers was calculated similarly, dividing potential nutrient recovery by projected fertilizer imports. Impacts specific to urban or rural areas were calculated using the same procedure as for total impacts.

As the input datasets differed in availability, the number of countries for which results have been reported varied across resources and sanitation categories, ranging from 117 to 159 countries (Figure S2). For example, impacts from populations served by newly-installed sanitation systems are reported for more countries than those from populations served by newly-treated wastewater systems, because the datasets related to untreated wastewater[71] included fewer countries. Results were only reported for countries with sufficient data (Figure S2), providing as much information as possible to guide decision-makers and/or future research, and revealing where current data gaps exist (Section S4).

Notably, due to the global nature of this analysis, contextual factors potentially hindering or accelerating implementation of resource recovery were not quantitatively considered because they are strongly dependent upon local context and specific system design[19,99–101,127]. These factors include: cultural norms, religious practices, and perceptions regarding human excreta; institutional and legal policies; economics; human health and perceived risks; environmental protection; and the priorities, knowledge, and resources of stakeholders, who should be involved in participatory and interdisciplinary frameworks to ensure sustainable local decision-making and appropriate technology choice around sanitation and resource flows[8,19,84,98,128–130]. As such, this analysis provides support and a quantitative foundation for future studies investigating the role and contextual relevance of factors such as these.

Regional and Global Summaries. Using results from countries with sufficient data, average impacts were calculated for geographic regions, defined according to SDG country groupings[104]. Subsequently, averages across development categories (least-developed, developing, developed) and the entire world were calculated. When calculating these averages, regional values were assigned to any countries without sufficient data (the number varies based on resource and sanitation category), ensuring different levels of data availability did not over- or under-represent any regions (Section S5).

Spatial Co-Location Analysis. Although nutrients recoverable from sanitation may exist in a country needing those nutrients, geographical disparities may result in costly transport, reducing the feasibility of reuse. To estimate the degree of spatial congruency between recoverable nutrients and nutrient requirements of agricultural crops, a spatial co-location analysis was performed. First, spatial distributions of crop nutrient requirements were estimated using global harvested areas of seventeen major crops from the year 2000 (5 arc minute × 5 arc minute cell resolution, or approximately 10 kilometers × 10 kilometers at the equator)[118] and median recommendations of crop-specific nutrient application rates (Table S11)[131]. Within each grid cell, the total nutrient requirements across all seventeen major crops were determined by summing the products of individual crop area and crop-specific nutrient recommendations. The total nutrient demand for each grid cell was then estimated by dividing the total nutrient requirements for the seventeen crops by the fraction of a country's total harvested crop area these seventeen crops represent[53]. Next, the global population distribution from the year 2000 (0.5 arc minute × 0.5 arc minute cell resolution, or approximately 1 kilometer × 1 kilometer at the equator)[119,120] was combined with calculated country-level per capita nutrient recovery potentials for 2000 to estimate the spatial distribution of recoverable nutrients, and cells were aggregated to match the resolution of crop nutrient recommendations. In each cell, recoverable nutrient quantities were compared with crop recommendations to identify co-located recoverable nutrients (i.e., recoverable nutrients not in excess of crop recommendations in the same cell). Finally, co-located recoverable nutrients and total recoverable nutrients (co-located and in excess) were independently summed in each country, calculating the country's co-location score as the fraction of total recoverable nutrients that are co-located. This procedure was performed separately for nitrogen, phosphorus, and potassium, and a weighted average across all three nutrients (similar to Eq. S12) provided a final aggregate score (Section S6).

Co-Location Sensitivity Analysis. As co-location scores were calculated for the year 2000 (due to data availability), it was uncertain whether comparisons between co-location scores and nutrient recovery potentials (projected for 2030) would be meaningful. Therefore, to test whether future changes in crop area or population could substantially alter country-level co-location, a sensitivity analysis was performed in which co-location scores were recalculated under three scenarios: (A) ±25% uniform change in crop nutrient requirements with no corresponding population change; (B) ±25% uniform change in population with no change in crop requirements; and (C) +50% increase in urban populations (defined as being located where population density is at least 150 people per square kilometer[132]) with no change in rural populations or crop needs. The final scenario represents extreme urbanization, in which all population growth from 2000 to 2030 occurs in urban areas (where nutrients are more likely to be in excess, decreasing co-location). Co-location scores for all countries exhibited only minor variations in response to the three scenarios (Figure S8). Although responses to the third scenario (C) were slightly larger, they remained relatively small (an average decrease of 5% across all countries, with a maximum decrease of 11%), suggesting each country's co-location score in 2030 will likely be similar to its score in 2000. Therefore, the sensitivity analysis indicates comparisons between co-location scores from 2000 and nutrient recovery potentials for 2030 (as in Figure 2.3) are likely to be meaningful, recognizing minor discrepancies in actual future co-location may occur (Section S7).

Recovery Technology Sensitivity Analysis. To assess the primary scenario's sensitivity to the use of different recovery technologies, the full analysis was repeated using distinct recovery efficiencies in centralized and decentralized systems. Urban and rural populations were assumed to use centralized and decentralized systems, respectively (except for rural sewer connections, classified as centralized). This analysis confirmed findings were not sensitive to different recovery efficiency assumptions in centralized versus decentralized systems (Figure S5).

Overall Uncertainty Analysis. Uncertainty surrounds several parameters related to nutrient and energy excretion, recovery efficiencies, system design life, and food waste (Table S5). Monte Carlo analysis with Latin Hypercube Sampling[133] (10,000 runs) was used to account for possible variations in input parameter values. The uncertainty analysis was performed for the primary scenario and the seven alternates (results presented in Figure S4), to ensure the primary scenario remained robust across potential variations. When presenting results below, the most probable value is shown first, followed by parenthetical ranges representing 5th and 95th percentile values from the distribution of outcomes produced by the uncertainty analysis around the primary scenario (for additional details see Section S8).

Results and Discussion

Global Impacts. If universal sanitation coverage is achieved by 2030 and the proportion of untreated wastewater is halved using nutrient recovery technologies, 15 (11-20) million metric tonnes (MMT) of nitrogen, 2.2 (1.1-3.4) MMT of phosphorus, and 4.0 (3.0-5.1) MMT of potassium could be recovered annually for agricultural use. These quantities represent 11% (9-16%), 9% (5-15%), and 12% (10-16%) of projected synthetic nitrogen, phosphorus, and potassium fertilizer use, respectively (Figure S6, Table S7). In contrast, energy recovery could impact global household electricity use by only 1% (0-2%). Recognizing household electricity merely represents a fraction of total energy use[106,125,126], the disparity between nutrient and energy recovery becomes even more dramatic, though not entirely unexpected given human metabolism. Fecal excretion accounts for only 2-10% of dietary energy intake, with most ingested energy being stored or oxidized to drive biological processes[50]. In contrast, adult nutrient ingestion and excretion are typically in balance[51,52]. Energy recovery from sanitation media currently receives greater attention than nutrient recovery[18,134], while a comparison of potential global nutrient and energy impacts suggests the opposite should be true.

Across all resources, recovery from newly-installed sanitation systems represents the majority (85-86%) of total potential impacts from meeting SDG Targets 6.2 and 6.3. Replacing existing systems with resource recovery technologies could further increase global potential to recover nutrients and energy by an additional 50-79%, suggesting resource recovery should be a primary focus as existing systems are replaced or upgraded. This point is particularly relevant in developed countries, where replacing existing systems will account for most (77-84%) of the total resource recovery potential and could offset 4-12% of these nations' projected synthetic fertilizer use (Figure S6; Table S7). As fertilizer prices are tied to demand, reduced synthetic fertilizer use in developed countries could affect the global market and improve fertilizer accessibility in developing countries[135]. Overall, across the three sanitation categories, newly-installed systems represent the greatest single contribution despite global sanitation coverage of 65% in 2010[103], reflecting the fact that growing populations[116] will require new installations along with those currently lacking access. Similarly, while global urban coverage (81%) was considerably higher than rural coverage (48%) in 2010[103], urban systems will account for most (57-60%) of the total impacts (Table S7) due to urbanization trends[116]: trends that can connect global urbanization with improved food security through nutrient and water reclamation[112].

Impacts in LDCs. While potential global impacts of nutrient recovery far surpass those of energy recovery, wide variations exist across countries and regions, with impacts from newly-installed sanitation systems highest among LDCs (Figure 2.1, Tables S7-S10). Aggregating across these nations, nutrient recovery from newly-installed systems could impact projected use of nitrogen, phosphorus, and potassium by 65% (55-94%), 68% (35-113%), and 149% (114-

193%), respectively, while energy recovery could impact use of household electricity by 17% (7-28%) (Figure 2.1). Impacts from newly-treated wastewater systems are negligible among most LDCs, due to the high cost and low use of conventional sewers[71,123]. Replacing existing systems could bring total impacts for nitrogen, phosphorus, potassium, and household electricity use to 84% (67-118%), 88% (42-141%), 193% (137-243%), and 22% (8-35%), respectively (Figure 2.1). Because LDCs typically exhibit low sanitation coverage[103] and high population growth[116], newly-installed systems account for the clear majority (77-83%) of potential impacts.

Among 31 LDCs with sufficient data to estimate nutrient and electricity impacts from resource recovery, nearly all (>90%) are characterized as having considerable resource limitations (falling within the shaded boxes in Figure S3), but relative impacts from newly-installed systems vary considerably (Figure 2.2; Table S8). Six nations could more than double or completely offset projected use (>100% impact) of both fertilizer nutrients and household electricity (Figure 2.2, Zone I). These countries may benefit from systems combining nutrient and energy recovery, perhaps separating nutrient-rich urine and energy-rich feces to simplify recovery processes[20,29,123]. Alternatively, a process that recovers energy from wastewater and then uses nutrient-rich, reclaimed water for irrigation may be advantageous in certain settings[136,137], although co-location (due to the energy intensity of transporting water) and matching embedded nutrient quantities with local crop needs[138] will be critical. Seventeen nations could double only nutrient use (Zone II) and may aim to focus on nutrient recovery. Alternatively, a combined system where recovered energy drives nutrient recovery or pathogen reduction components[124] may also be effective. One country (Mali) can double household electricity but is unlikely to double nutrient use (Zone III). A focus on energy recovery may be warranted, although the form and use of recovered energy (e.g., methane for cooking, heat for pathogen reduction) should depend on local needs. Finally, while seven countries are unlikely to double either nutrient or household electricity use (Zone IV), meaningful improvements may yet be generated. Every setting should consider specific context when determining appropriate options, but it may be most crucial in Zone IV to ensure potential benefits are consequential.

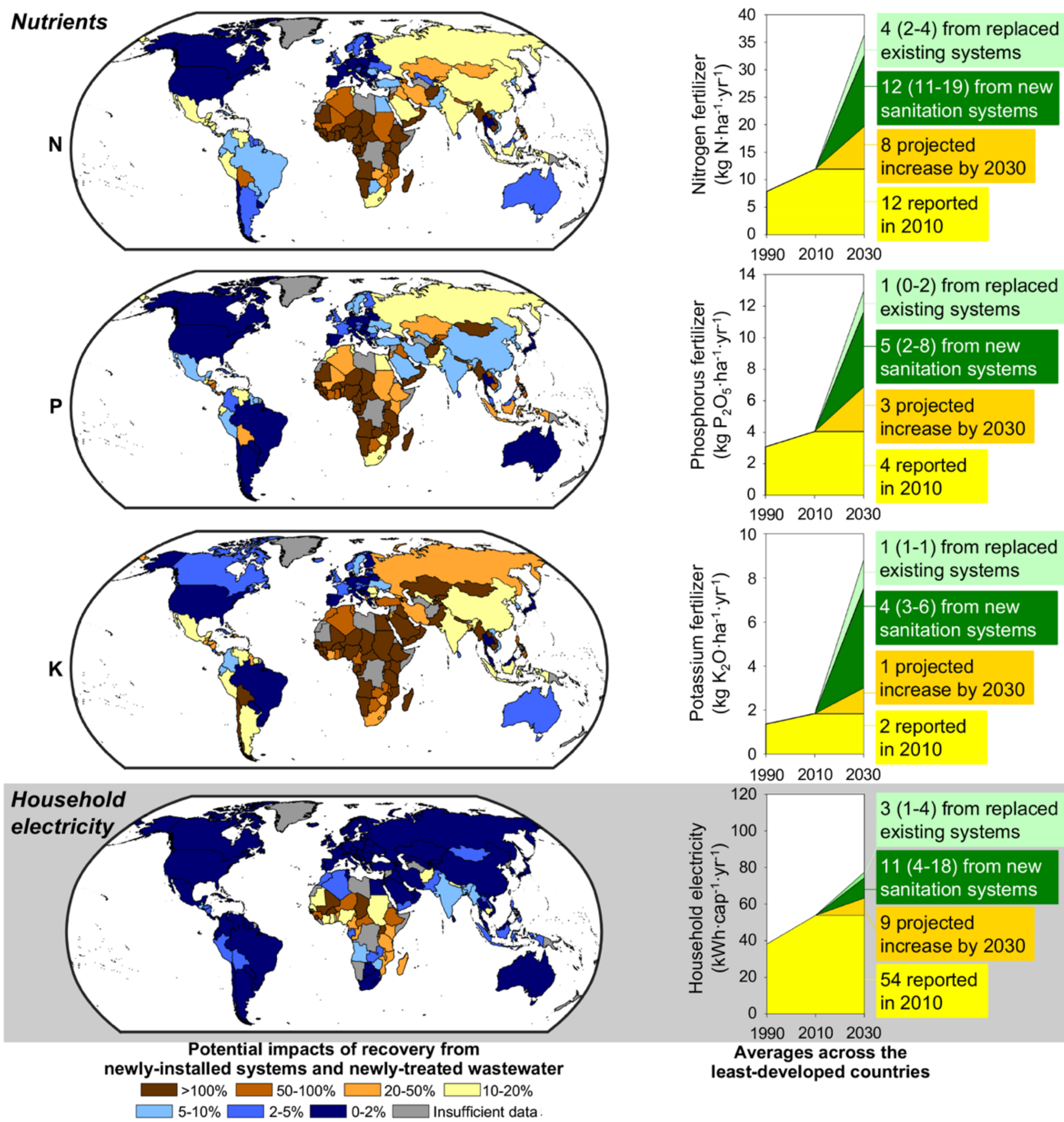


Figure 2.1. Potential impacts of resource recovery from sanitation in 2030. Impacts on projected consumption of fertilizer nutrients (per hectare cropland) and household electricity (per capita) in 2030 (the SDG time horizon). Maps on the left show country-by-country impacts on nutrient and household electricity access that could stem from newly-installed sanitation systems and new wastewater treatment capable of resource recovery, normalized to projected 2030 consumption. A 100% increase (dark brown shading), therefore, means that projected 2030 levels of that resource could double or all projected resource use could be replaced if resource recovery technologies were used to achieve SDG Targets 6.2 and 6.3. Graphs on the right show aggregate results for all least-developed countries. Impacts from newly-treated wastewater systems are negligible and not annotated, while additional increases from existing system replacement are shown. The boxed annotations on the right provide expected values for the segment of the plot with the corresponding color (reported value of resource use in 2010; projected increase in resource use by 2030; further potential impacts from newly-installed sanitation systems or replaced existing systems), and parenthetical ranges represent 5th and 95th percentile values from the uncertainty analysis around potential impacts from resource recovery. Regional averages for all three sanitation categories are presented graphically in Figure S6 and numerically in Table S7, and individual country results are presented in Tables S8-S10. Maps were created using Matlab 2015a.

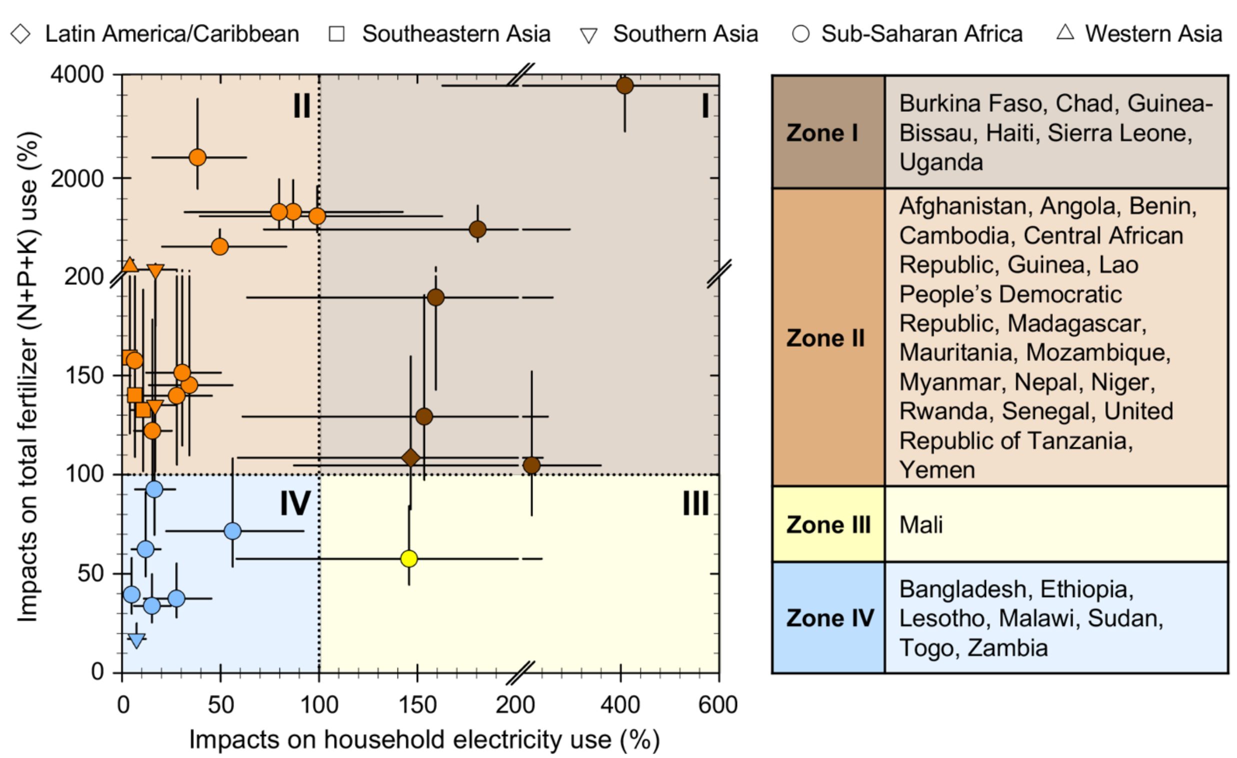


Figure 2.2. Potential impacts of resource recovery from newly-installed systems on the use of fertilizer nutrients (per hectare of cropland) and household electricity (per capita) for 31 least-developed countries in 2030. Error bars show 5th and 95th percentile values from the uncertainty analysis. Diagonal lines crossing each axis indicate a scale change. Countries are grouped according to geographic region (symbol shape) and plot position (zone and color). Countries in Zone I (>100% impact on both axes) have the potential to either double use of both resources or completely offset unsustainable sources of nutrients and energy (supplementing projected use may be most likely for resource-limited settings in LDCs). Nations in Zone II could realize impacts on nutrient use of at least 100%, while those in Zone III are associated with household electricity impacts of at least 100%. Although countries in Zone IV are unlikely to reach similarly high impact levels for either resource, substantial impacts may still be achievable. Numerical results for newly-installed systems in all countries are presented in Table S8.

Additionally, impacts on individual nutrients are not necessarily uniform among LDCs (Tables S8-S10). Certain countries (e.g., Guinea-Bissau) would benefit more greatly from nitrogen recovery, while others (e.g., Zambia) can realize more significant impacts from phosphorus recovery. Still others (e.g., Sudan) would find potassium recovery to be most impactful for fertilizer access. Potassium fertilizer availability is especially limited in many sub-Saharan African countries[53,121] due to a scarcity of potash ores within the region and high transportation costs that hinder trade[139]. Accordingly, the potential impacts of potassium recovery are particularly high in these countries, despite little attention devoted to the development of effective technologies[62]. The potassium estimates presented here reflect recovery solely through direct reuse of source-separated urine (which contains most excreted nutrients), while an additional 6-23% of excreted potassium would be available in feces[54,140]. Overall, despite variations, many LDCs (45% of those with sufficient data) can double supplies of all three nutrients, suggesting combined nutrient recovery technologies (e.g., low-cost urine separation[29,123]) would be especially useful. Moreover, as recovery leads to greater nutrient availability in resource-limited settings, crop productivity will likely improve[31,141], increasing food supplies. In turn, improved diets would lead to larger quantities of resources in excreta. Although this effect was left out of this analysis to avoid introducing additional uncertainty, nutrient reuse can create a positive feedback loop, generating recoverable resources at levels greater than those predicted here in locations where they are needed most.

Recovering nutrients in places with limited fertilizer access could also mitigate equity concerns[30]. In contrast with current circumstances, in which crucial resources (e.g., phosphate rock, potassium ore) are regionally concentrated[139,142] and difficult to obtain for those facing geographic and economic constraints, resource recovery from sanitation could distribute materials more equally. Excreted nutrients are available wherever humans live, although ensuring equitable access to safe technologies remains a significant hurdle[143]. Perhaps the possibility for reduced reliance on nutrient imports will foster progress toward developing safe and accessible systems. For multiple countries across the economic spectrum, widespread nutrient recovery could replace meaningful percentages of synthetic fertilizer imports (Figure S7), reducing the need for international transfers that require considerable energy and imperfectly balance uneven distributions of non-renewable minerals. For example, Canada and Kazakhstan could completely replace their need for potassium fertilizer imports through potassium recovery from replaced existing systems, while Nicaragua could replace all imported phosphorus fertilizers with recovered phosphorus from newly-installed systems.

Additionally, given that high regional transportation costs present a barrier to access, recovering nutrients in rural areas, where large portions of many LDC populations reside[116] (Figure S3(C)), may pose fewer logistical challenges than opportunities presented by urban recovery and subsequent transport to rural agricultural land[112] if crops are far from population centers. Indeed, more than half (60-62%) of LDCs' potential impacts from resource recovery could be generated in rural areas (Table S7). Additionally, rural areas may present more feasible opportunities to recycle organic waste from humans and animals through onsite systems and nearby agricultural land. This strategy can improve nutrient levels, although fully closing yield gaps may require additional local fertilizer inputs[144] (access to which might be increased through nutrient recycling in developed countries[135]). An additional benefit of organic waste recycling is the replenishment of soil organic matter, which may be depleted due to prolonged low-input cultivation (commonly practiced in LDCs). Depletion of organic matter can lead to issues such as accelerated soil erosion, declining structural stability, and diminished yields[144,145]. Therefore, decentralized approaches (also potentially appropriate in rapidly growing urban centers[123]) could be critical in fully realizing sanitation's potential to address multiple SDGs, although considering economies of scale, recovery efficiencies, and system density[146] will be essential to evaluate tradeoffs and financial viability when collecting resources from small onsite systems in sparsely populated areas.

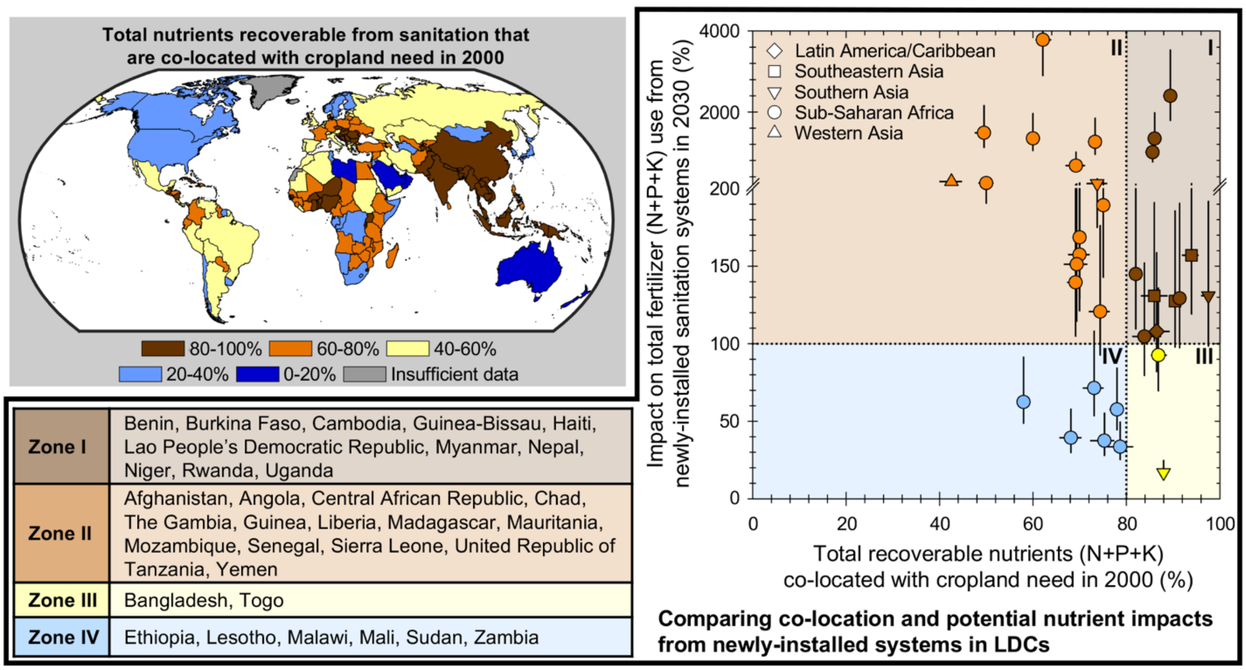
Regarding energy recovery, while this analysis compared potential generation of useable electricity with projected household electricity use, many families rely on other energy sources for various needs. Notably, low household electricity use is often coupled with heavy reliance on solid cooking fuels (Figure S3(B)). Using solid fuels (e.g., firewood, charcoal) can contribute to deforestation, loss of productive time, and poor indoor air quality leading to increased disease burden[147–149]. Simultaneously, however, traditional solid fuels may connect to important social traditions (e.g., family members gathering at the fireplace in the evening[150]), implying that systems recovering and using energy from sanitation must be culturally appropriate. Energy recovered from sanitation could be used to partially meet a household's cooking needs, provide heating for the home or to sanitize human excreta, or generate lighting[124,151]. For example, a household recovering energy from human sanitation in sub-Saharan Africa could meet approximately 10%

of its cooking needs with a biogas stove, thereby reducing solid fuel use by a similar percentage (assumptions and calculations in Section S9). At the other extreme, one person's excreta could offset less than 1% of an average resident's use in the United States (10 MWh per capita per year of household energy in 2009[152]), suggesting household-level energy recovery would be insignificant in developed nations.

Nutrient Co-Location. In both rural and urban areas, the feasibility of nutrient recovery may also be affected by proximity to application sites. Analyzing the spatial distributions of recoverable nutrients and crop requirements revealed co-location was decidedly regional, with countries in South and Southeastern Asia, Eastern Europe, Latin America, and sub-Saharan Africa (especially West Africa) exhibiting high scores (Figure 2.3). Additionally, the co-location analysis confirmed many LDCs exhibit the potential to form meaningful connections between sanitation and agriculture with minimal transport requirements. In thirteen of 33 LDCs with sufficient data to analyze co-location, at least 80% of recoverable nutrients are co-located with crop requirements (Figure 2.3, Zones I and III), and eleven of these thirteen countries also have the potential to double or completely offset fertilizer use through resources recovered from newly-installed systems (Zone I). Co-location scores of several additional LDCs fall just below 80%.

The energy required to recover and transport urine has been compared with the energy needed to produce and distribute synthetic fertilizers in locations such as Sweden and the United States. These studies report favorable energetics of urine recovery if transport distances remain below a location-specific threshold (40-220 kilometers)[110,127,153], suggesting countries with large percentages of co-located nutrients could develop highly efficient agricultural systems through nutrient recovery. Even for other nations (including developed countries) with lower scores, the reported transport distances suggest nutrient reuse may remain viable. However, accurately estimating transport requirements, considering systems that integrate livestock manure, defining distance thresholds, and developing appropriate recommendations for nutrient recycling is dependent on multiple context-specific factors, and further study is needed at various scales[76,129].

Notably, most small island nations could not be included in this analysis because spatial crop area data were insufficient. Further study is needed for Small Island Developing States, which are typically characterized by high population densities, limited land resources, and high transportation and infrastructure costs[154].



Zone I	Benin, Burkina Faso, Cambodia, Guinea-Bissau, Haiti, Lao People's Democratic Republic, Myanmar, Nepal, Niger, Rwanda, Uganda
Zone II	Afghanistan, Angola, Central African Republic, Chad, The Gambia, Guinea, Liberia, Madagascar, Mauritania, Mozambique, Senegal, Sierra Leone, United Republic of Tanzania, Yemen
Zone III	Bangladesh, Togo
Zone IV	Ethiopia, Lesotho, Malawi, Mali, Sudan, Zambia

Figure 2.3. Co-location of nutrients recoverable from human populations and crop requirements. The global map (upper left) presents the fraction of each country's recoverable nutrients that are located within the same 5 arc minute × 5 arc minute grid cell (approximately 10 km × 10 km at the equator) as crops requiring those nutrients. Any nutrients in excess of crop needs are not counted as being co-located. Co-location scores were calculated using the most recent available spatial datasets (from 2000). The results of a sensitivity analysis (Figure S8) show country-level scores to be insensitive to possible future changes, suggesting co-location scores should be similar in 2030. The remainder of the figure compares co-location scores with potential nutrient impacts from newly-installed sanitation systems in 33 LDCs. In the scatter plot, error bars indicate 5th and 95th percentile values from the uncertainty analysis, and diagonal lines crossing the y-axis indicate a scale change. Countries are grouped according to geographic region (symbol shape) and position on the plot (zone and color). Numerical co-location scores for all countries are presented in Table S12. Maps were created using Matlab 2015a.

A Foundation Supporting Focused Efforts. Overall, this study's results support previous work suggesting efforts surrounding resource recovery from sanitation could most effectively improve agricultural nutrient access and food security in resource-limited settings by focusing on new systems that will be installed to achieve universal coverage[29,155]. Simultaneously, recovered resources from replaced existing systems provide opportunities for offsetting synthetic fertilizer use in developed countries. Systems in both urban and rural areas can play an important role. Urban areas are growing and represent larger potential for global resource increases, especially in developed countries (where urban areas account for 80-81% of potential impacts). However, LDCs will continue to be predominantly rural until 2030 and will likely require decentralized options to achieve universal sanitation coverage, although urbanization is projected to continue after 2030[116]. Regardless of scale, in many LDCs, these systems are likely to be highly co-located with crop needs, a further incentive to capitalize on potential connections between sanitation and agriculture.

While this analysis reveals several overarching trends that could guide resource recovery efforts in both resource-limited and highly-industrialized settings, certain countries do not follow the patterns common among others with similar levels of development, appearing to align more closely with distinct geographic regions. For example, counter to general trends observed in Figure S3(A), Bangladesh, India, and Pakistan (all located in South Asia and formerly parts of British colonial India) are characterized by relatively low sanitation coverage (<60%)[103] but moderately high fertilizer consumption (>130 $kg \cdot ha^{-1}$ $N+P_2O_5+K_2O$)[53]. A possible explanation is that agriculture is crucial to these countries' efforts to feed their large populations (with cropland covering over 50% of total land area)[53]. As such, potential impacts of nutrient recovery are relatively low, even for Bangladesh (a LDC). However, as these countries already use considerable quantities of synthetic fertilizer, some use could be offset with recovered nutrients (e.g., Bangladesh could replace 10-33% of projected phosphorus fertilizer use through recovery from newly-installed systems). By leveraging the datasets developed here, future work to investigate local drivers and conditions in specific contexts could generate detailed insights and recommendations applicable in those settings.

These findings can also function as a point of departure for future work exploring other key context-specific factors (e.g., cultural norms, perceived health risks, institutional policies, economics) that may hinder or accelerate resource recovery. Such considerations were not integrated into the quantitative analysis because they are strongly dependent upon local setting and specific system design[19,99–101,127]. Ensuring recovery systems are safe, accessible to all segments of the population, equitably owned and managed, and cost-effective when compared with conventional options will be critical. Additional analyses occurring within specific contextual settings may help to overcome common barriers to sustainable development and sanitation success, including political instability, low willingness to undertake large infrastructure projects, and limited household income for implementing safely managed systems[123,156]: this is especially true for work on decentralized systems associated with less initial capital investment, less complex infrastructure, and low-technology resource recovery mechanisms. Improving education in LDCs and incorporating resource recovery concepts into technical curricula can also help to train a new generation ready to design contextually-appropriate systems[156] and prevent the apparent and hidden failures continuing to plague water and sanitation infrastructure in developing countries[37].

Economics, Energetics, and Markets. Across various contexts, understanding economic aspects will be particularly critical to ensuring the long-term feasibility of resource recovery efforts. While some global estimates of funds needed to achieve universal sanitation coverage have been developed for specific technologies (with resource recovery systems such as urine-diverting dry toilets comparing favorably against conventional wastewater treatment)[8], these estimates are highly uncertain, and true costs are sensitive to contextual features including local markets, regulatory frameworks, geography, and population density[8,99–101,127]. Case studies in cities across sub-Saharan Africa, Sweden, and the United States have shown, in certain circumstances, the economics, energy requirements, and market demand associated with alternative systems and recovered products may be similar to or better than conventional wastewater treatment and fertilizer production[100,101,110,127]. Conversely, in a case study assessing struvite recovery from urine in Nepal, current circumstances (e.g., fertilizer prices, magnesium costs) prevented the system from becoming financially sustainable[157]. From a more general energetic perspective, nitrogen fixation through the Haber-Bosch process uses 37 $MJ \cdot kg\ N^{-1}$, while an additional 12-45 MJ·kg N^{-1} is required for wastewater treatment to convert reduced nitrogen back to N_2 (via nitrification/denitrification or mainstream deammonification)[99]. Recovering nitrogen through thermal volume reduction of collected urine compares favorably (34 MJ·kg N^{-1}), whereas other processes such as ion exchange (116 MJ·kg N^{-1}) are less efficient[99].

Without further innovation, even appropriately-designed resource recovery systems are likely to entail higher initial capital costs than the most basic conventional approaches (e.g., pit latrines)[158,159]. They may require innovative management models[123] to capitalize on potential revenues and creatively utilize alternative funding mechanisms (e.g., related to carbon offsets and nutrients embedded in reclaimed wastewater[136,137]), providing long-term financial incentives that economically justify their implementation and reduce barriers to sustained adoption[101].

Limitations. While this analysis integrated elements not often considered in high-level global assessments (e.g., household-level food waste, spatial co-location within countries, data uncertainty), it must be interpreted in light of certain limitations. First, the analysis of resource recovery impacts employed the same recovery efficiencies for all categories of sanitation systems, while in reality different technologies and degrees of centralization will not provide the same level of treatment. Due to uncertainty surrounding current and future technology implementation, this analysis focused on diet-based per capita recovery, scaled up according to populations projected to be served by each sanitation category and compared with projected resource use, incorporating a diverse set of technology options when defining recovery efficiency ranges for the uncertainty analysis. Second, this analysis quantitatively compared energy recovery with household electricity use due to incomplete data on total household energy use, and because the impacts with respect to total national energy use were inconsequential. In reality, recovered energy can be used in other forms and for other purposes than to provide household-level electricity. Third, several factors dependent upon specific context and technology choice (e.g., recovery costs, regulatory policies, stakeholder perceptions, cultural norms) could not be quantitatively incorporated into this global-level analysis.

The co-location analysis was limited by available spatial data, with the most recent crop area datasets existing for the year 2000 and lacking data in some smaller nations (especially Small Island Developing States). Additionally, the co-location score was based on the data's cell resolution rather than on distance thresholds that would define favorable nutrient reuse (as these thresholds are highly dependent on local setting). In many cases, this study's limitations are related to data gaps and context-specific factors, suggesting where further study may be most useful.

Portfolios of Approaches. While this study supports previous findings that nutrients from excreta could replace meaningful percentages of synthetic fertilizers globally[29–33], potential impacts of energy recovery are much less significant, and recovered resources are insufficient to fully address either nutrient or energy needs. Therefore, resource recovery from sanitation should function as one feature within a larger portfolio of approaches promoting sustainable resource flows. For example, lessening synthetic fertilizer dependence will also require reducing inefficiencies in food production and distribution (e.g., minimizing nutrient runoff, shifting away from meat-intensive diets, decreasing supply chain losses) and recovering nutrients from other organic materials (e.g., animal manure)[51,144,160,161].

Within the sanitation sector itself, a diverse portfolio of technologies, management systems, and decision-making frameworks is needed to account for local drivers. The findings presented here suggest incorporating resource recovery into sanitation systems provides an opportunity to simultaneously benefit multiple societal goals, including universal sanitation coverage (SDG Target 6.2), improved access to nutrients (SDG 2) and household energy (SDG 7), and greater sustainability of global nutrient cycles. However, realizing the full benefits of resource recovery will require future work investigating specific contexts, identifying locally appropriate connections across multiple goals, and exploring questions of feasibility. The datasets produced in this study provide a quantitative foundation for local work that must continue in various settings around the world, re-envisioning sanitation, integrating multidimensional goals, and providing appropriate, context-specific technology choices. Rather than being considered an unmentionable and resource-intensive burden, sanitation could become an inspirational component of societal infrastructure, acting as an amplifying force for sustainable development.

CHAPTER 3: RECIRCULATION OF HUMAN-DERIVED NUTRIENTS FROM CITIES TO AGRICULTURE ACROSS SIX CONTINENTS[b]

Introduction

The global population continues to rise and require ever-greater quantities of food[160], while urbanization has increasingly separated food production and consumption[162]. Nutrient flows through rural agricultural production and urban food and water systems could enhance or jeopardize future global sustainability, as manifested in contexts including urban metabolism studies[163,164], the circular economy[165,166], and the United Nations' Sustainable Development Goals (SDGs)[12]. Over the past century, synthetic fertilizers have supplied agricultural nutrients to enable dramatic expansions in food production[30,66]. However, phosphorus and potassium fertilizer production depends upon regionally-concentrated, non-renewable supplies of phosphate rock and potash ores[30,67], while converting atmospheric nitrogen gas into ammonia fertilizer through the Haber-Bosch process is energy-intensive. Upon reaching farmland, excessive fertilizer application can lead to water contamination, algal blooms, and eutrophication[1].

Alternatively, nutrients in human excreta, most of which are not currently recycled to cropland (e.g., <15% of excreted nitrogen)[4,31,109], could offset globally meaningful quantities of synthetic fertilizer use and advance food security goals by increasing nutrient access in low-income countries[29,167]. National strategies in countries such as Rwanda are promoting resource recovery[168], while global agencies advocate for sustainable water and sanitation systems, urban environments, and consumption patterns (SDGs 6, 11, 12)[12]. Simultaneously, international research collaborations estimate anthropogenic nutrient flows already exceed planetary

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boundaries and assert these grand challenges require systems-wide transformations (including nutrient recycling)[3,38,169].

Urban areas represent critical contexts for nutrient recovery, food security, and sustainability. They can act as either centers of innovation or focal points for environmental deterioration[170]. Urban settings now contain over 50% of the global population (including nearly 500 cities with populations >1 million and 28 megacities with populations >10 million)[116], and they may house 6.3 billion by 2050[116]. As urban agriculture (limited by available land) accounts for a small fraction of urban diets[75], cities rely on rural food production[162,170], transporting in nutrients for consumption. Rather than allowing these nutrients to pass through urban metabolisms and pollute local environments[4], recovery from urban sanitation represents a potential link to close nutrient cycles between cities and rural agriculture[171]. From local businesses finding profits from human waste in Kigali and Accra[172] to Ostara's 17 struvite recovery installations serving 11.5 million people across North America and Europe[173], planners and entrepreneurs are experimenting with systems to capitalize on the resources in urban sanitation.

However, while various recovery technologies have been developed and implemented[18,174], roadmaps to help cities make informed decisions are often nonexistent, particularly regarding the challenge of finding markets for recovered products[101,175]. For cities, the distances between nutrient recovery and agriculture are a key factor in these markets[76], potentially constraining recovery technology choice and feasibility due to transport energy requirements. Long transport distances may make reuse of products with relatively low nutrient content (e.g., reclaimed wastewater) prohibitively expensive[76], placing greater pressure on cities to recover highly-concentrated products (e.g., struvite) using more complex processes. Therefore, there is a critical need to characterize the co-location of urban nutrients with cropland where they can be applied. However, beyond findings specific to certain crops, locations, or nutrients (e.g., 74% of United States corn's phosphorus demand could be met using in-county recyclable sources)[76], studies analyzing nutrient co-location across diverse urban areas are lacking[170].

These analyses would provide valuable information surrounding the feasibility of nutrient recycling, elucidating strategies for harmonizing urban wastewater management with agricultural needs[138].

Accordingly, we undertook an exploratory exercise to characterize the spatial co-location of cropland and recoverable human-derived nutrients from major urban centers to define paths forward for closing urban nutrient cycles. For 56 of the world's largest urban agglomerations (hereafter referred to as cities) across six continents (Supplementary Tables 1-2), we defined urban extents and estimated total nitrogen, phosphorus, and potassium quantities recoverable from human excreta (typically the largest source of nutrients entering urban sanitation[4,34]; Supplementary Methods 1, Supplementary Table 3). Using crop-specific fertilizer recommendations (Supplementary Table 4), we then characterized the distances nutrients must travel to satisfy crop demands (Supplementary Methods 2; Supplementary Figures 1-3). Additionally, we assessed sensitivity to changing nutrient supplies and multiple sanitation infrastructure configurations (Supplementary Table 5; sensitivity of recoverable nutrient quantities to recovery efficiency has been evaluated previously[167]). Furthermore, to guide decision-makers, we estimated (i) how shifts in existing crop patterns can reduce distances (Supplementary Table 5) and (ii) how recovering more concentrated nutrient products can reduce transport energy requirements. Ultimately, this exercise enables us to identify broad patterns and locations where nutrient recovery strategies and products may be most able to advance the circular economy, creating opportunities for future context-specific studies to promote nutrient recycling and boost rural productivity through cooperation between urban water and regional agriculture systems.

Methods

Agricultural Nutrient Requirements. Spatial distributions of crop nutrient requirements were estimated using global harvested areas of crops from the year 2000 (5×5 arc-minute cell resolution, or approximately 10×10 kilometers at the equator)[118] and median recommended nitrogen, phosphorus, and potassium application rates for each crop (Supplementary Table 4)[131]. Fifty-two crops were considered to be nationally significant, defined here as constituting at least 10% of the total crop area in any given country (determined using FAO data[53]), and were directly included in the calculations that produced the overall distribution of agricultural nutrient requirements. Within each grid cell, the total nitrogen, phosphorus, and potassium requirements across all 52 crops were determined by summing the products of individual crop area and crop-specific nutrient requirements. The total nutrient demands for each grid cell were then estimated by dividing these values (the total N, P, and K requirements for the 52 crops) by the fraction of a country's total harvested crop area attributed to these 52 crops[53]. The 52 nationally significant crops accounted for 90% of all cropland area, both globally and across the 31 countries containing the 56 cities included in our analysis. However, it is worth noting that some nutrient-intensive crops not classified as nationally significant (e.g., specialty fruits and vegetables) may be grown around cities, potentially reducing estimated nutrient distances.

City Extents and Recoverable Nutrients. To ensure the analysis included cities from geographically diverse regions, the ten most populous cities from each inhabited continent (based on total population in the year 2000, as reported in the UN World Urbanization Prospects[116]) were selected from a spatial database of large urban areas[116,176]. As the database includes only six urban areas for Oceania (i.e., Australia and proximate islands)[177], the selection process yielded a total of 56 geographically, economically, and ecologically heterogeneous cities (Supplementary Table 1).

The spatial extent of each city (i.e., the area containing all city residents) was defined as a contiguous area containing the city's spatial coordinates[116,176] and meeting or exceeding a given population density threshold (Supplementary Figure 1). As cities in different parts of the world are characterized by a wide variety of population density levels and spatial attributes, each city's density threshold was identified individually. Using global datasets of population distribution in 2000 (0.5 $\times$ 0.5 arc-minute cell resolution, or approximately 1 $\times$ 1 kilometer at the equator)[119,120], contiguous areas were defined for a large range of population density threshold values (2-10,000 people per square kilometer). For each threshold value, the total population contained within the defined area was calculated and compared with the city's population figure (as reported in the UN World Urbanization Prospects[116]), and the threshold value producing the minimum difference in total population was selected. This procedure resulted in a range of density threshold values across the 56 cities (Supplementary Table 2), reflecting the fact that these cities are laid out in various ways, from those which are densely packed to those with considerable urban sprawl (Supplementary Figure 3). Each city's defined extent was also used to estimate city area and average population density (Supplementary Table 2).

Finally, the population distribution within each city's extent was aggregated to match the cell resolution of crop nutrient requirements. Populations were multiplied by country-level per capita nutrient recovery potentials from human excreta in 2000 to arrive at a spatial distribution of human-derived nutrients recoverable from sanitation within the city (human excreta are estimated to be the largest nutrient source in urban sanitation)[4,34]. Recovery potentials were estimated from information on per capita protein and caloric supply, nutrient excretion, and recovery efficiencies (accounting for most nutrient losses in the recovery process) across various technology options (e.g., solid precipitation, adsorption, ion exchange, ammonia stripping, direct reuse of source-separated urine and feces) using procedures from previous work (Supplementary Methods 1, Supplementary Table 3)[167].

Nutrient Distance Analysis. Prior to conducting the nutrient distance analysis, datasets for each city were converted from the geographic coordinate system into a projected Universal Transverse Mercator (UTM) coordinate system locally appropriate to each city to enable more accurate distance calculations (Supplementary Table 2 shows selected UTM zones for each city). To correct for any discrepancies in total city population and any errors resulting from the altered cell sizes and orientations introduced when converting between coordinate systems, recoverable nutrient data were scaled to ensure the total nutrients in the city's extent agreed with the population figure reported by the UN[116].

We assumed nutrients recovered from a given city would not be allowed to cross national borders to reach cropland in a different country. Therefore, the agricultural nitrogen, phosphorus, and potassium requirements data were clipped to include only the country where the city resides, after which these datasets were also projected into the appropriate UTM coordinate system.

As agricultural nutrients are required in various ratios depending on the crops being grown, and as ratios typically differ from those available in human excreta, three individual nutrient distance analyses (nitrogen, phosphorus, and potassium) were conducted for each city. In each analysis, an iterative process allocated recoverable nutrients from the city to the closest cropland demanding those nutrients, ensuring that crop requirements were not exceeded (Supplementary Methods 2; Supplementary Figure 2). To begin each iteration, path distances were calculated from any cells with nonzero agricultural nutrient requirements to all other cells in the country. This operation accounted for changes in elevation using a global elevation raster (GTOPO30, 0.5×0.5 arc-minute cell resolution, aggregated and projected into the local UTM coordinate system)[178] and only considered overland travel (i.e., travel could not occur directly through water bodies) using a global land area mask (0.5×0.5 arc-minute cell resolution, aggregated and projected)[179]. Road networks were not considered, because spatial data on global road networks are of highly variable quality across countries containing the cities in our analysis (Supplementary Table 13). However, to investigate how road networks might affect our distance estimates, we conducted a quality control analysis for all 56 cities, choosing five cropland locations and comparing our distance estimates from the city center with road distances obtained after inputting the coordinates into Google Maps. Each cropland location was randomly selected from a grid showing cells with potential nutrient application and containing no information on roads, and distances were measured to the centroid of the cell. On average, constraining transport to roadways was observed to affect distance measurements for a given city by 7-21% (Supplementary Methods 2;

Supplementary Tables 14-15). These differences will not change the broad trends observed in average distances, which span approximately two orders of magnitude across the 56 cities. In practice, locations and quality of transport infrastructure will play an important role in the feasibility of nutrient reuse, and should be considered when developing more precise estimates at the local level.

Following the path distance operation, we identified the city cell that was closest to a cell demanding nutrients, and a quantity of nutrients was transferred from the city cell to the cropland cell. If the cell's total quantity of recoverable nutrients fell below what was required by the cropland, all nutrients were moved, whereas only enough nutrients to fully meet crop requirements were transferred if recoverable nutrients exceeded the demand. This procedure was repeated until all recoverable nutrients had been moved to cropland. The alternate condition, in which all cropland was saturated with nutrients (i.e., the country's nutrient demands were fully met before the city's recoverable nutrients were exhausted), was also a possible scenario to complete the iterative process, but this condition was never satisfied.

To complete the analysis for one nutrient, results from all iterations were merged, defining the full agricultural area that could be fertilized by that nutrient through recovery from sanitation in a given city (Supplementary Figure 3). These results indicated the nutrient quantities applied in each cell and the distances those quantities needed to travel. We calculated the city's total nutrient mass, the distances required to utilize specified mass fractions of nutrients (5%, 25%, 50%, 75%, 95%), and a mass-weighted average distance of complete (100%) nutrient application. Each city's total mass of recoverable nitrogen, phosphorus, and potassium were also compared with annual fertilizer imports into the country from 2000 to 2010, to determine whether nutrient recovery from a nation's largest cities could substantially reduce reliance on foreign fertilizer supplies. The procedure summarized above constituted the primary nutrient distance scenario (see Supplementary Methods 2 for further details), and the analysis was repeated under three altered scenarios (*centralized*, *increased population/affluence*, and *altered crops*; described below and in

Supplementary Table 5) to test the sensitivity of results and assess the potential impact of shifts in local crop types.

Sensitivity Analyses. The preceding nutrient distance analysis was repeated twice to assess the sensitivity of the primary scenario's results to changes in various conditions. In brief, the two sensitivity analyses included: (i) altered locations of recovered nutrients within cities to reflect high centralization of sanitation systems; and (ii) altered estimates to reflect potential increases in city population and food supply. These two scenarios are described in detail below.

The first alternate scenario (*centralized*) acknowledges that the nutrient distance analysis relies on a certain procedure for defining city extents (i.e., a contiguous area meeting or exceeding a given population density threshold) and identifying where recoverable nutrients are located (based on population density within the city extents). However, urban extents are notoriously difficult to delineate, and a variety of definitions are in use that incorporate diverse factors (e.g., population density, economic criteria, the presence of human-made structures)[74,75,132]. Additionally, the primary analysis assumes nutrients from human waste can be recovered in the grid cell where they are generated. Stated differently, sanitation and nutrient recovery systems are assumed to be somewhat decentralized, with recovery occurring in each 10 $\times$ 10-kilometer grid cell within the city. Depending on the city's degree of urban sprawl, this assumption of decentralized systems could substantially impact nutrient distances. Therefore, the centralized scenario assumed all recoverable nutrients from each city's population were concentrated in a single location (i.e., each city's sanitation system was fully centralized, with all nutrient recovery occurring at one point). This location was assumed to be in the center of the city (defined according to the city's latitude and longitude as reported by the UN[116]), which in most cases would place recovered nutrients at the largest possible distance from surrounding cropland. Repeating the analysis in this way characterizes the sensitivity of results to the definitions of urban extents and the degree of sanitation system centralization.

The second alternate scenario (*increased population/affluence*) considered the potential for rising city populations and food supplies. City populations from the year 2000 were used in the original analysis to correspond with harvested crop area datasets. However, populations of some cities have already shifted dramatically since 2000, and they are projected to continue changing in the future[116]. We characterized the increased population/affluence scenario using city population estimates for 2030[116], along with estimated per capita nutrient recovery potentials from human excreta in 2030 (to account for changes in nutrient excretion due to changing food supplies; Supplementary Methods 1)[167]. The new populations were distributed throughout each city's urban extent by scaling up the density distributions from 2000. As this scenario is meant to assess sensitivity to increased supplies of recoverable nutrients (rather than provide a complete picture of future conditions), it neglects potential expansions in urban area, shifts in relative population density within a city's extent, and land use changes near the city. However, these additional changes may be substantial (and interrelated)[72,180], and future context-specific studies geared toward individual city planning should account for their potential impacts.

Altered Crop Patterns. An additional scenario (*altered crops*) evaluated whether nutrients' travel distances could be substantially reduced through changes in local crop patterns. For each country containing at least one of the 56 cities, nationally significant crops (accounting for >10% of that country's total harvested crop area) were considered as possibilities that could be grown around cities. The nationally significant crops with the highest nitrogen, phosphorus, and potassium requirements were independently identified, and existing crops were replaced wherever a grid cell's nutrient demand was lower than it would be by growing the selected nationally significant crop. This process occurred separately for each nutrient, so that optimal scenarios for nitrogen, phosphorus, and potassium could each be considered. As such, this scenario provides relevant information regarding how crop type could support local nutrient recovery and reuse in agriculture.

Nutrient Recovery Products Analysis. In addition to the distance nutrients must travel, the form in which they are recovered also plays a key role in developing efficient reuse systems. Depending on technology and process choice, nutrient recovery can generate numerous products of varying composition, ranging from dilute nutrients in treated wastewater to more nutrient-dense crystal products (e.g., ammonium sulfate, ammonium struvite, potassium struvite)[62,157,181–183]. Each product's nutrient concentration will determine the total mass that must be transported to deliver a given quantity of nutrients to cropland. While the same transport distance (average distance from the primary scenario) is used for each product in this exercise regardless of concentration, transport energy can be reduced substantially when nutrients are in more concentrated forms (e.g., crystal products). Therefore, we identified multiple recovery products (treated wastewater without nutrient removal, dewatered sludge from anaerobic digestion, undiluted urine, crystal products [ammonium sulfate, ammonium struvite, potassium struvite]) to evaluate. Using reported compositions of each product (accounting for typical values and possible variations)[5,20,182,183], we estimated the total mass needed to deliver one tonne of each nutrient (N, P, K). The products' mass factors (i.e., mass of product per mass of nutrient) were multiplied by representative distances for each city (i.e., that city's average nutrient distances from the primary scenario) and energy factors (i.e., required transport energy per mass and distance transported) to estimate the energy required to transport one tonne of each nutrient to cropland. Transport energy calculations assumed reclaimed wastewater was pumped to cropland (pumping energy estimated using equations and assumptions from Shoener et al.[16]), while other products traveled by truck (road freight vehicle energy estimated using the ecoinvent 2.2 database). In addition to variations in product composition, reported energy ranges also reflect a wide spectrum of energy values from different pumping velocities and freight vehicles (all parameter ranges provided in Supplementary Table 12).

These results were compared with estimates of the energy required for synthetic fertilizer production and transport (Figure 3.3), calculated using literature data[26] and the ecoinvent 2.2

database, accounting for production of various single-nutrient fertilizers (urea, ammonium nitrate, ammonium sulfate, triple superphosphate, single superphosphate, potassium chloride, potassium sulfate). The comparison provides an assessment of whether transport of recovered nutrients using a certain product in a given city may require less energy than synthetic fertilizer production and distribution.

Statistical Analyses. After completing the nutrient distance analysis, mass-weighted average distances and the distances required to utilize specified mass fractions of each recoverable nutrient were compared with various city characteristics to identify trends and correlations. Spearman's rank-order correlation coefficients and p-values (two-tailed)[184] were calculated for each pairing of nutrient distances and quantitative city characteristics (including: average cropland density within 10, 50, and 100 kilometers of city boundaries[185]; total city population[116]; city area and average population density [calculated using the defined city extents]; total recoverable nutrients [calculated from city population[116] and per capita nutrient recovery potentials][167]; country per capita GDP in 2000)[186]. We also performed simple linear regressions to provide a rough indication of the magnitude of a given factor's effects, complementing the Spearman's correlation analysis. Kruskal-Wallis tests (i.e., one-way ANOVA on ranks, a non-parametric method for data that are not normally distributed)[187] were performed for each combination of nutrient distances and categorical city characteristics (including: continent; whether the city is located near a coast [i.e., within 100 kilometers[188] of an ocean or major freshwater body]). We also computed simple averages and standard deviations for selected categorical groupings to complement the Kruskal-Wallis tests. All statistical calculations were performed in Matlab R2015a.

Results and Discussion

Nutrient Travel Distances in 2000. Based on city population distributions, crop patterns, and nutrient recovery potentials from sanitation in the year 2000 (primary scenario), travel distances to apply nutrients to cropland vary widely across 56 cities (Figure 3.1; Supplementary Table 6). Average nitrogen distances (i.e., average distance per kilogram for complete application of all recoverable nitrogen; Eq. S4) span approximately two orders of magnitude, ranging from six kilometers (Rome) to 329 kilometers (Boston). Moreover, among some cities in Brazil (Rio de Janeiro, Sao Paulo), Japan (Osaka-Kobe, Tokyo), and the United States (Boston, New York-Newark, Philadelphia, Washington, D.C.), application areas from multiple cities overlap, suggesting their distances may be longer if recovery is broadly pursued.

Higher local cropland density (i.e., cropland relative to land area) was the factor most associated with shorter average distances (average cropland density was computed within 10, 50, and 100 kilometers of city boundaries; in all cases, $p<0.0001$; Supplementary Table 7). As a rough indication of this factor's impact, a simple linear regression estimated a 1% increase in cropland density within 50 kilometers of city boundaries correlates with a decrease in average nitrogen distance of 1.6 kilometers (95% confidence intervals: 1.0-2.2 kilometers; Supplementary Table 8). Disparities in distances also reveal that cities in Europe, Africa, and Asia typically exhibit shorter distances than cities on other continents (e.g., $p<0.001$ for nitrogen; mean of average nitrogen distances across cities in Europe, Africa, and Asia: 40 kilometers; mean in South America, Oceania, and North America: 91 kilometers; Supplementary Tables 9-10). These differences likely connect to variations in cropland density. The mean cropland density within 50 kilometers of cities in Europe, Africa, and Asia is more than twice that of cities on other continents ($p<0.001$).

Beyond cropland density, nutrient distances may relate to additional factors, including crop nutrient demand ratios, population density, city area, coasts, city population and food supply, and a country's per capita GDP. Despite large variations across cities, potassium distances are nearly always shortest (followed by phosphorus and then nitrogen), due to imbalances between the ratios of nutrients required by many crops and the typical ratios in human excreta. Compared with many crops' recommended N:P:K application ratios (e.g., 1:0.25:1.27 for wheat; Supplementary

Table 4), typical ratios of nutrients excreted by humans are higher in nitrogen and lower in phosphorus and potassium (global average of 1:0.14:0.29; although processes such as anaerobic digestion can result in substantial nitrogen losses, altering ratios in recovered products)[20,167]. Therefore, crop nitrogen demand in a given location is often met first (so that nitrogen must travel farther), while a given cropland area can often absorb a greater fraction of human-derived potassium.

While distance variations seem most directly related to the prevalence and type of local crops, they may also be partly associated with population density and city area (Supplementary Figure 3). Cities with larger areas and smaller population densities, suggesting urban sprawl[189], are associated with lower cropland densities within 50 kilometers of city boundaries ($p<0.001$). City area also relates to the sensitivity of nutrient distances to assumptions regarding sanitation system configuration. Our primary results assumed sanitation systems were distributed throughout each city in a somewhat decentralized configuration, with one system per 100-km^2 grid cell. If all nutrients are instead recovered at one centralized location (centralized scenario), distances to cropland almost always increase (Supplementary Figure 4). The magnitude of this change is positively associated with city area ($p<0.001$), as nutrients centrally collected must often travel farther to reach rural cropland. However, these increases do not fundamentally alter overarching patterns broadly evident across cities.

Coastal cities (i.e., within 100 kilometers[188] of oceans or major freshwater bodies) may exhibit longer average distances (p=0.01-0.03 across all nutrients, Supplementary Table 9), as nutrient movement is constrained by coastlines. When analyzing each continent individually, coastal characteristics may play a role in Asia (p=0.02-0.05), where the longest distances are associated with coastal cities (e.g., Tokyo, Osaka-Kobe, Karachi). Overall, however, coasts appear to be less critical than continental differences, with non-coastal cities in the Americas (e.g., Mexico City, Belo Horizonte) still exhibiting longer distances than most African, Asian, and European cities. Local cropland density again appears to factor prominently, as differences in the average cropland densities of coastal and noncoastal cities were minor (p=0.15), while larger disparities were observed across continents.

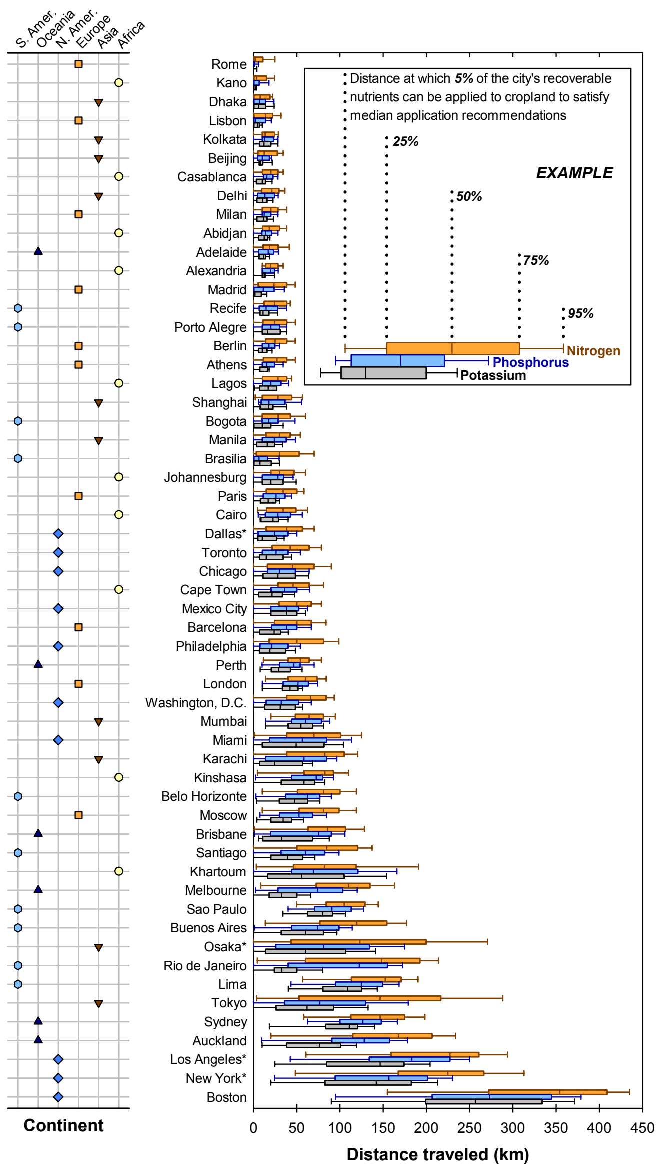

Figure 3.1. Distributions of nutrient transport distances for 56 cities in 2000. Distributions of nitrogen, phosphorus, and potassium travel distances for 56 cities across six continents in 2000, ordered by average nitrogen distances (i.e., average distance per kilogram for complete application of all recoverable nitrogen; Eq. S4). As shown in the inset example (upper right), each box-and-whisker distribution shows the distance various mass fractions of a city's total recoverable nutrients (5% [left end of whisker], 25% [left edge of box], 50% [median line in box], 75% [right edge of box], 95% [right end of whisker]) must travel to be fully applied. All distances are from the primary scenario. The continent of each city is shown on the left. Supplementary Table 6 provides all numerical results. * Multi-city urban agglomerations (Dallas-Fort Worth, Los-Angeles-Long Beach-Santa Ana, New York-Newark, Osaka-Kobe) appear under a single city name to reduce figure width.

The Role of City Population. Key in determining nutrient distance is a city's total population (Figure 3.2 for nitrogen; Supplementary Figures 5-6 for phosphorus and potassium).

With more people come greater nutrient quantities more likely to saturate local cropland. For example, Tokyo and Osaka-Kobe are associated with long distances partly because they are two of the world’s most populous urban agglomerations. Inversely, smaller cities such as Lisbon, Kano, and Abidjan exhibit short distances.

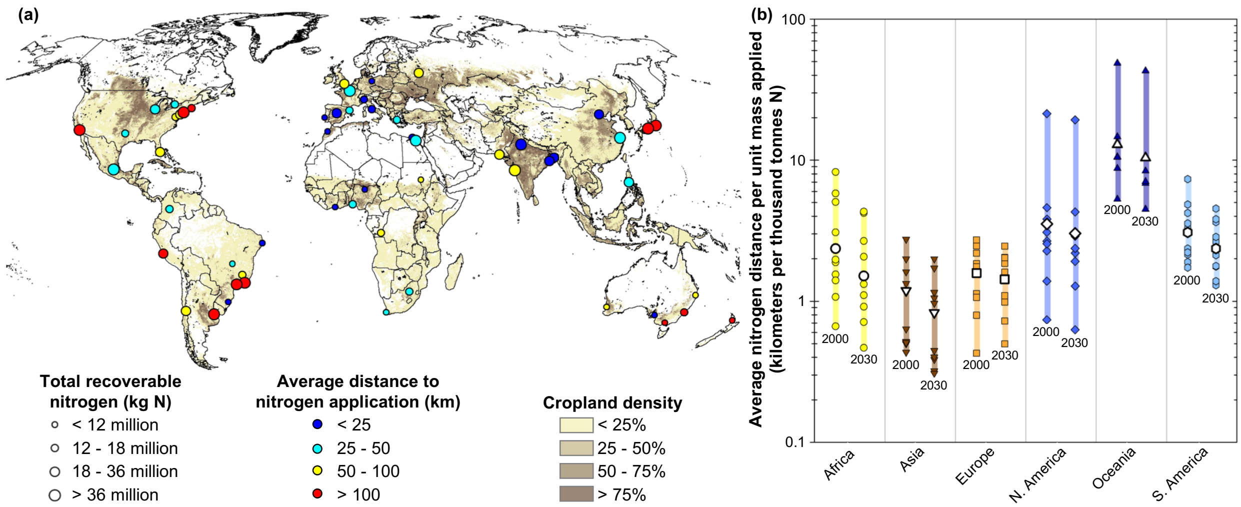

Figure 3.2. Recoverable nitrogen quantities and average transport distances. Total recoverable nitrogen quantities compared with mass-weighted average nitrogen distances (for 100% application to cropland) for all cities. Each point on the left map **(a)** shows results from the primary scenario (in 2000). The right plot **(b)** shows each city’s average nitrogen distance per thousand tonnes of nitrogen applied, grouped by continent. For each continent, the left vertical grouping shows values from the primary scenario (labeled “2000”), while the right grouping shows results from the increased population/affluence scenario (labeled “2030” for simplicity). Larger white symbols represent mass-weighted averages for each continent (i.e., average distance per thousand tonnes of nitrogen applied from all cities in that continent, weighted according to each city’s total recoverable nitrogen). Phosphorus and potassium results, which follow similar trends, are displayed in Supplementary Figures 5-6.

This pattern does not always hold, however, and continental grouping still appears to play a strong role (Figure 3.2). Many African, European, and especially Asian cities exhibit short distances relative to total nutrient quantity, while most Oceanic cities have exceptionally long distances for their small sizes. The four least populous cities are all in Oceania, but, except for Adelaide, they exhibit some of the longest nutrient distances. Inversely, larger Asian cities (e.g., Dhaka, Kolkata) tend to be associated with shorter distances, as they are often surrounded by high densities of nutrient-intensive crops. Although Asia’s two most populous cities in 2000 (Tokyo, Osaka-Kobe) exhibit long distances, their average distances per thousand tonnes of nutrient applied are comparable with the rest of Asia and much lower than those of other smaller cities (e.g., Auckland, Boston). These trends suggest reuse may be beneficial among some larger cities, even when average distances are not particularly low (e.g., Mexico City). Indeed, the association between distance and population is relatively weak (e.g., $p=0.04$ for nitrogen; Supplementary Table 7) compared with the impact of cropland density ($p<0.001$). Along with cropland density, identifying contexts most conducive to reuse will likely also depend on the types of crops being grown, affected by local characteristics such as climate, topography, and soil quality.

At national scales, recoverable nutrients from some large cities could replace sizeable fractions of fertilizer imports, although global fertilizer trade can be volatile and may change with large-scale nutrient recovery. Examining data from 2000 to 2010[53], Cairo's recoverable nutrients could have offset all of Egypt's annual phosphorus fertilizer imports and 23-70% of imported potassium (Egypt's net food imports [i.e., imports less exports] were 9-20% of total food supply by mass; Supplementary Table 11). In Japan (where net food imports represented 46-49% of food supply), Tokyo and Osaka-Kobe together could have replaced >72% of nitrogen fertilizer imports, while recovery in Buenos Aires could have offset >25% of potassium fertilizer imports in Argentina (a net food exporter). Smaller cities in low-income nations may provide a similar function. On average, recovery in Khartoum could have offset 73% of potassium fertilizers imported by Sudan (where net food imports represented 3-11% of food supply), although complete reuse would require long travel distances. Given the substantial yield gaps observed in sub-Saharan Africa[144], however, recovered nutrients may function to supplement (rather than offset) imported fertilizers.

Rising populations and food supplies (considered to 2030 in the increased population/affluence scenario; other factors including urban extent and land use are held constant to assess sensitivity specific to changing nutrient supplies) will greatly increase recoverable nutrients in many cities, particularly in Africa and Asia (except Japan, due to low growth projections). At the most extreme, nitrogen quantities in Kinshasa and Lagos are projected to quadruple from 2000 to 2030 (based on population and food supply projections[116,121];

Supplementary Methods 1). However, average distances per mass of nutrient typically decrease or remain similar to values from the primary scenario (Figure 3.2). While distances must rise to accommodate larger nutrient quantities, distance typically increases to a lesser degree than quantity, as cropland density often intensifies further from cities. In Africa and Asia, with rapidly growing urban populations and substantial agriculture near cities, reductions in distance per unit mass are especially pronounced (36% average reduction in Africa, 30% in Asia), suggesting nutrient reuse may become increasingly efficient with rising population and affluence. In contrast, average distance per unit mass decreases by about 10% in Europe, where smaller population/affluence changes are expected. However, this sensitivity analysis is limited in that it does not consider changes in urban extent or land use. By 2030, urban expansion may displace approximately 2% of global cropland, but local impacts could be more substantial around certain cities in Africa and Asia (e.g., Alexandria, Kolkata)[180], affecting reuse possibilities and necessitating adaptive decision-making.

Locations interested in exploring nutrient recycling may consider approaches to reduce nutrient transport distances and energy requirements. Below, we consider possibilities related to (i) altering local crop patterns and (ii) implementing sanitation technologies that generate concentrated nutrient products.

Impacts of Altering Local Crop Patterns. Crop type is critical in determining nutrient demands of surrounding cropland, and growing more nutrient-intensive options could reduce nutrient distances. Acknowledging that crop choice should consider several factors involving climate, soil, topography, economics, and other local conditions, we performed a hypothetical exercise to assess the impact of altering crop patterns to optimize for nutrient distance. Essentially, we evaluated the sensitivity of nutrient distances to crop type. Various crops could be chosen (e.g., nutrient-intensive vegetables, commonly grown in urban agriculture due to their perishability and nutritional benefits[75]; Supplementary Table 4), but we limited our analysis to each country's nationally significant crops (i.e., already grown on >10% of cropland), assuming these crops may represent viable alternatives for farmers. We replaced existing crops with the most nutrient-intensive nationally significant crop wherever it would increase nutrient demand (Supplementary Table 5). As N:P:K ratios differ, a specific crop was selected to optimize for each nutrient in the country.

Depending on existing and available replacement crops, some cities could dramatically reduce nutrient distances (Supplementary Figure 7). In southern Europe, many cities could reduce already short distances by replacing local crops with olives, which demand high inputs of all three nutrients. For example, Rome could reduce average nitrogen distance up to 76% if olives replaced all existing local crops. Plantains (nationally significant in Colombia) also demand high levels of all three nutrients. Consequently, Bogota could reduce average distances up to 47% (N), 66% (P), and 80% (K). In Nigeria, nationally significant crops are limited to sorghum and millet, which are not particularly nutrient-intensive. However, as sorghum demands more nitrogen and potassium, Kano's nutrient distances could be reduced by replacing locally-grown millet with sorghum. Lagos, where oil palm (nutrient-intensive but covering <10% of Nigeria's cropland) is common, would not see similar benefits. Likewise, shifting crop patterns where nutrient-intensive, nationally significant crops are already common (e.g., rice around Dhaka, Osaka-Kobe, and Tokyo) would not create meaningful change.

These findings should be interpreted with caution, as altering crop patterns to reduce nutrient distance may conflict with other important factors and could be inadvisable. These factors include: economics (e.g., shifting to more nutrient-intensive crops may be less profitable than maintaining current practices); food security (e.g., shifting to cotton around Karachi may reduce food access in Pakistan, where 44% of children are stunted[190]); resource-efficient crop rotation systems (e.g., maize-soy rotations near Chicago should not be abandoned to grow only maize); tensions between farmer-level concerns (e.g., economic and resource productivity) and system-level issues (e.g., water and land footprints)[191]; and local climate, soil, and topographic conditions. Thus, improving conditions for nutrient recycling may require other approaches.

Impacts of Nutrient Recovery Products. Recovering nutrients in more concentrated forms may also increase reuse feasibility by reducing the mass that must be transported a given distance (thereby reducing energy requirements). Depending upon the sanitation technologies employed, nutrients are recovered in products of varying composition (Figure 3.3). Along with travel distance, each product's nutrient content determines whether it represents an energetically (and, to an extent, economically) favorable alternative. For simplicity, we compared transport with different products over each city's average nutrient distances (primary scenario) as a representative case. Using each product's nutrient content and the energy consumption of road freight vehicles (or pumping, for recovered wastewater; Supplementary Table 12), distances were converted to transportation energy estimates (to visualize general trends, Figure 3.3 shows ranges encompassing all cities). Furthermore, estimating global energy demands of synthetic fertilizer production and distribution offers a point of comparison (Figure 3.3).

Wastewater treated to reduce organic matter and pathogens (but not nutrients) can irrigate cropland ("fertigation," providing combined nutrient and water recovery, particularly useful where water limitations constrain crop production)[171] but is relatively dilute[5]. Although pumping consumes far less energy than trucking per unit mass and distance, recovered wastewater requires more energy than other products because nutrients constitute <0.01% of its total mass (Figure 3.3). Additionally, as its large volume precludes storage, wastewater application typically must occur immediately after treatment and delivery. In contrast, dewatered sludge (15-28% solids) has higher nutrient concentrations[5] (but also higher nitrogen losses during digestion, not considered in our recovery potentials) and could be stored until appropriate application times. Sludge provides opportunities particularly for phosphorus reuse, while being less competitive for potassium (Supplementary Figure 8). Urine is especially nitrogen-rich[20], but it also provides opportunities for other nutrients. However, urine reuse may be infeasible where conventional sewer infrastructure is in place (resulting in mixed waste streams).

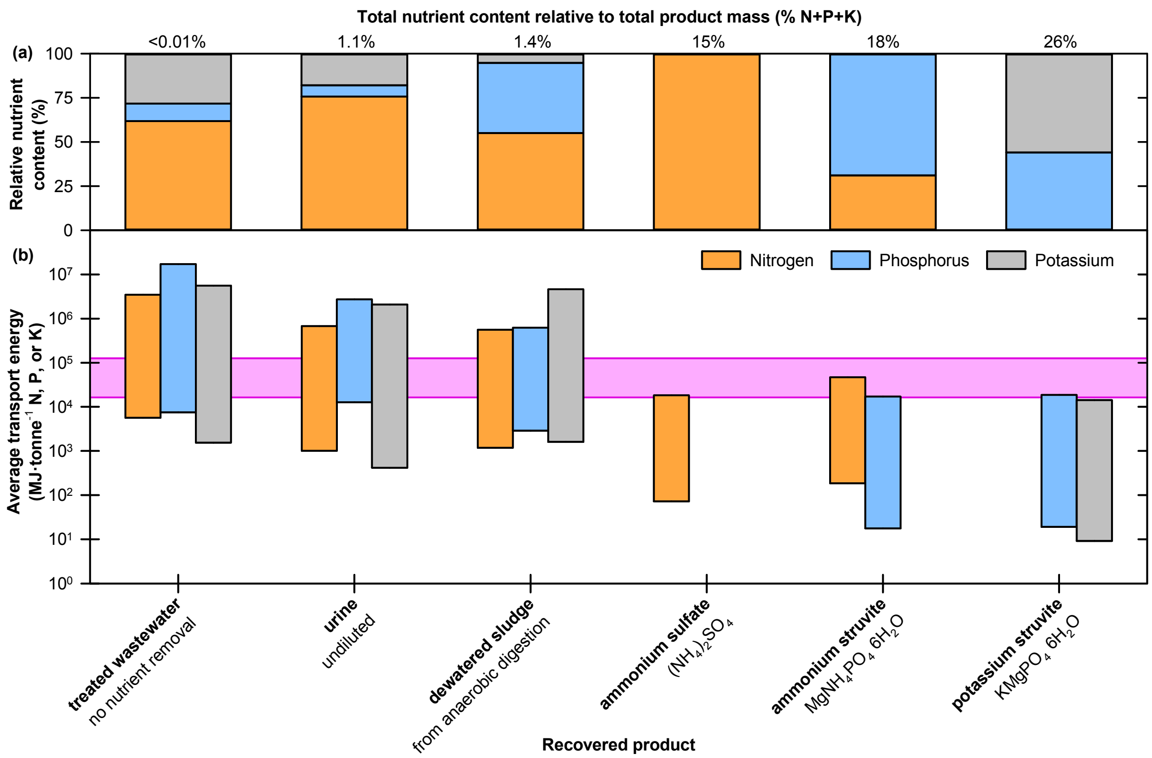

Figure 3.3. The impact of recovery product on transport energy requirements. Transport energy per tonne of nutrient applied using different recovery products across all cities. The relative fractions of nitrogen, phosphorus, and potassium typically contained in each product are shown in the upper graph **(a)**, with total nutrient content (N+P+K) relative to each product's total mass represented by the percentages along the top. In the lower graph **(b)**, each bar represents the range of energy values across all 56 cities, calculated from representative distances (each city's average distance for each nutrient), nutrient concentrations in recovery products (mass of N, P, or K per total mass), and estimates of transport energy per tonne-kilometer. The full range accounts for uncertainty around each product's nutrient composition and transport energy requirements (minimum to maximum value, inclusive of uncertainty for all 56 cities). The light-purple region extending horizontally through the lower graph shows the range of estimated energy demands for production and transport across all single-nutrient synthetic fertilizers (global averages), allowing for comparisons between nutrient reuse and synthetic fertilizers. Supplementary Figure 8 shows individual results for each city. Supplementary Table 12 provides details regarding the nutrient composition and energy values used in the calculations.

Crystal products – dried solids including ammonium sulfate, ammonium struvite, and potassium struvite[157,181,182] – represent the most concentrated form of recoverable nutrients from mixed waste streams, enabling storage and more distant transport. Most common is ammonium struvite ($MgNH_4PO_4 \cdot 6H_2O$), a mineral precipitate containing high phosphorus levels but no potassium[157], while ammonium sulfate [$(NH_4)_2SO_4$] and potassium struvite ($KMgPO_4 \cdot 6H_2O$) enable concentrated nitrogen and potassium recovery. For all crystal products, expected transport energies fall below global averages of synthetic fertilizer production and distribution, even among cities with long distances (Supplementary Figure 8), suggesting these locations may still benefit energetically from agricultural reuse if recovery of highly-concentrated products is feasible. However, crystal products' economic viability may be limited in certain locations (e.g., high magnesium costs in Nepal have hindered ammonium struvite systems[157]), and they require additional energy for recovery (e.g., roughly 21 $MJ \cdot kg\ P^{-1}$ for ammonium struvite precipitation[192], which is substantial relative to transport energy but would not push any cities past the phosphorus fertilizer threshold). Generally, choosing appropriate technologies and products will depend on capital and operating expenses (e.g., precipitation reactors, chemical addition), existing sanitation infrastructure (e.g., level of centralization, conveyance systems), treatment process configuration (e.g., existence of sidestreams including anaerobic digester supernatant), and wastewater and sidestream composition (e.g., potassium struvite precipitation must follow ammonium oxidation)[5,157,182]. Furthermore, technology selection should occur in collaboration with farmers, ensuring end users will accept and value recovered products[101].

Limitations. The results of this exercise should be taken as first-order estimates of nutrient transport distances across 56 cities, useful for identifying broad trends and locations that may warrant further investigation into reuse strategies. Various limitations suggest possible avenues for future research that will improve accuracy and prioritize opportunities to link urban and agricultural metabolisms. First, the primary scenario is based on distributions of population, crop demands, and estimated nutrient recovery potentials from 2000. Each dataset is associated with uncertainty, and many cities have expanded considerably since 2000. We also assumed that nutrients could not cross national borders, but international transfers may be relevant for metropolitan regions near or extending across boundaries (e.g., Tijuana-San Diego, East Africa's Lake Victoria region). Furthermore, this analysis approximated urban sanitation facility locations, with one facility per 100-km^2 area. We challenged this assumption by repeating the exercise with one facility per city. Some cities were sensitive to facility location, but general patterns remained consistent (Supplementary Figure 4). Each city's true sanitation network likely falls between these two bounding scenarios. Globally, while many cities have centralized infrastructure, nearly 30%

of global urban residents use onsite sanitation systems (potentially associated with sludge collection, transport, and semi-centralized treatment)[69]. Aspirationally, the exercise assumes complete sanitation coverage in cities, but achievable near-term nutrient recovery will depend on spatially explicit sanitation access, which can vary widely within subnational regions due to wealth and other factors[69]. Additionally, the definition of urban extents remains uncertain, especially for cities with growing populations. Generally, these limitations reflect the need for more accurate, temporally resolved data and definitional clarity on urban extent and infrastructure[75,193] to better estimate the characteristics and requirements of nutrient reuse in future, context-specific studies.

Implications. This analysis studied trends across a diverse set of 56 large urban agglomerations, identifying locations where recirculation of human-derived nutrients may be spatially feasible and considering strategies to reduce transport distance and energy. It shows universal promotion of agricultural nutrient reuse may sometimes be impractical (though concerns such as eutrophication provide alternative support for nutrient recovery). Rather, settings with characteristics including high local cropland density and compact urban area (e.g., Alexandria, Dhaka, Kolkata, Kano) should be identified and assessed for their potential to optimize nutrient reuse. Here, we have considered nutrient recovery and agricultural reuse as an additional process connected with existing or future sanitation treatment, estimating transport requirements of moving recovered nutrients to cropland. Our findings should be complemented with place-based studies able to holistically consider specific systems, and they offer a starting point for policy-makers, funding agencies, agricultural researchers and practitioners, development professionals, and utilities. Overarching patterns regarding reuse feasibility, intervention strategies, and the magnitude of potential opportunities and challenges provide insight to layer on top of locality-specific decision-making processes that consider current sanitation infrastructure, regulations, energy and labor requirements, and local agriculture.

Recycling nutrients from human sanitation can reduce global reliance on synthetic fertilizers and provide greater nutrient access in resource-limited settings[167]. The world contains at least two billion smallholder farmers typically living in lower-income countries[180], with yields often constrained by nutrient and water limitations[144]. Many smallholders live in Africa and Asia[194], two of the three continents (along with Europe) typically containing cities with shorter nutrient distances (Figures 3.1-3.2; although cropland in Africa and Asia may be more vulnerable to future urban land expansion)[180]. Lower per capita GDP is associated with shorter average distances (p=0.002-0.02; Supplementary Table 7), suggesting opportunities to support smallholder farmers through nutrient reuse. Where feasible, increasing linkages between cities and rural cropland by recirculating human-derived nutrients could improve farmers' economic and food security, reduce urban discharges to the environment, increase national food system resilience against international fertilizer and food price spikes, and motivate sanitation improvements where access is limited[4,34,171,180].

CHAPTER 4: ALIGNING PRODUCT CHEMISTRY AND SOIL CONTEXT FOR AGRONOMIC REUSE OF HUMAN-DERIVED RESOURCES[c]

Introduction

Nutrient inputs are needed to meet the agricultural productivity requirements of a growing global population and replenish nutrient export associated with harvested crops or environmental transport. The past century's use of inorganic inputs (e.g., Haber-Bosch nitrogen, mined phosphate rock) has enabled dramatic increases in food production[30,195,196] but has caused substantial environmental degradation (e.g., eutrophication). Discharges of anthropogenically-mobilized phosphorus and anthropogenically-fixed reactive nitrogen already exceed estimated planetary boundaries, beyond which abrupt global system shifts may occur[3,64]. Additionally, converting atmospheric nitrogen gas into ammonia fertilizer through the Haber-Bosch process is energy-intensive[66], while phosphorus and potassium fertilizers are produced from finite, geographically-concentrated supplies of phosphate rock and potassium ores[30,67].

Global nutrient flows through agricultural systems and human populations are characterized by substantial losses, including urban and agricultural runoff and leaching (leading to eutrophication), gaseous nitrogen emissions (e.g., nitrous oxide, a potent greenhouse gas), and food supply chain losses[31,35,197]. Concurrently, many farmers in resource-limited settings face low fertilizer access, constraining regional food production and food security[29,53]. For example, much of Uganda's cropland is nutrient-limited[144], but national surveys suggest only 3.2% of Ugandan farming households use fertilizers[198] (reflecting factors such as the prohibitive cost of imported nutrients, limited supplies, credit constraints, and poor transportation networks[199]).

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Nutrient waste streams (e.g., human or animal waste collected in sanitation or manure management systems) represent recoverable flows that could alleviate regional limitations on nutrient access and offset sizeable fractions of global fertilizer consumption[167] (Figure S1, Tables S1-S3). Based on existing literature estimates (used to develop Figure S1), over half of the nitrogen in livestock manure is already recycled, while recirculation of human-derived nitrogen remains relatively limited (<15% of nitrogen in human excreta is recycled)[4,31,109]. If all unrecovered human-derived nitrogen could be recovered and recirculated, it could offset 16-21% of inorganic nitrogen inputs to agriculture. A greater portion of human-derived phosphorus (<55%) is estimated to be recycled[107,109], likely because sewage sludge (with high phosphorus but low nitrogen levels due to gaseous nitrogen removal during processing[200]) is a common source of recycled nutrients. Unrecovered human-derived phosphorus could offset 9-12% of inorganic phosphorus inputs to agriculture. Moreover, as these waste flows are often collected and aggregated, they may be easier to capture than more diffuse flows (e.g., agricultural runoff). Other alternative nutrient sources (e.g., phosphorus from animal bone products) may offer further recycling opportunities[201].

However, this global mass balance assumes an idealized, homogeneous world, where wastes can easily be reused. In reality, regions are heterogeneous, characterized by variations in population density, sanitation access, crop/livestock systems, climate, topography, soils, and other factors. Previous research has, for example, estimated distances human-derived nutrients produced in urban settings would need to travel for cropland application, finding wide variations (spanning two orders of magnitude) across 56 of the world's largest cities[202]. Moreover, local soil conditions may play a particularly important role in determining whether resource recovery is worth pursuing or even possible. In a given soil context, different types of nutrient products (e.g., reclaimed wastewater, digested sludge, compost, source-separated urine, crystalline products) will behave differently from one another once applied to the field and may have divergent impacts on crop production, nutrient use efficiency, and soil quality[77–83]. However, little work has been done on a global level to assess and compare the suitability of potential recovery products relative to soil conditions. Thus, soil context could play an important role in driving decisions around whether nutrient recovery should be pursued and what recovery products should be generated and/or reused in a given locality.

The objective of this work was to assess the soil suitability of various human-derived nutrient recovery products on a global scale. We evaluated recovery products based on their suitability to soil context and used global soil data[203] to generate soil suitability maps for each product. These maps can help frame and guide conversations that consider local soil conditions when making decisions around nutrient recovery, identifying locations where certain products may be detrimental or where they may improve existing conditions. Further, we consider relationships between the potential magnitude of recovered nutrients if products are reused locally (acknowledging products may also be exported or transported in-country to appropriate reuse locations) and the soil suitability of specific recovery products. We discuss how this information might inform decision-making and investment to simultaneously advance Sustainable Development Goals for sanitation and food security[12]. Overall, this global study offers a foundation for incorporating soil suitability into analyses and discussions surrounding locally appropriate sanitation, nutrient recovery, and agricultural reuse.

Methods

Recovery products and pathways. Sanitation systems can employ various pathways to generate numerous products for nutrient recovery. In our analysis, we evaluated seven products, including reclaimed wastewater, digested sludge, compost, source-separated urine, ammonium sulfate, ammonium struvite, and potassium struvite (Tables S4-S5 show relationships between product characteristics and various soil parameters). For wastewater, we considered two global treatment and recovery cases: aerobic (conventional activated sludge) or anaerobic (upflow anaerobic sludge blanket) treatment without biological nutrient removal[5], allowing most nutrients to remain in the reclaimed effluent. Urine, containing most of the nutrients humans excrete[20,48], was assumed to be source-separated and stored in closed containers for treatment (minimizing ammonia volatilization)[61]. Solid products rich in organic carbon include anaerobically digested sludge and aerobically treated compost (we assumed both were generated from source-separated feces)[5,151,204–206]. Crystalline products, including ammonium sulfate ($(NH_4)_2SO_4$), ammonium struvite ($MgNH_4PO_4 \cdot 6H_2O$), and potassium struvite ($KMgPO_4 \cdot 6H_2O$), are nutrient-dense materials recovered through processes such as precipitation (struvite) or ammonia stripping and absorption (ammonium sulfate)[181–183], each of which may require substantial quantities of chemical additives and energy-intensive separation techniques[207,208]. We assumed crystalline products were recovered from source-separated urine (a more nutrient-dense stream than domestic wastewater).

Soil suitability parameters. We evaluated these recovery products relative to spatially-explicit soil parameters. In this global analysis, we considered five parameters that may impact whether application of nutrient recovery products is locally suitable (pH, sodicity, clay content, soil cation exchange capacity [CEC], and clay fraction CEC; Tables S6-S7), acknowledging that numerous additional factors will also play a role in many specific cases. Our selection of parameters and threshold levels was based on a literature review focused on soil classifications and fertility in agricultural settings (Section S1). Given that suitable soil conditions will vary depending on local factors such as climate and crop selection, we define uncertainty ranges for each parameter rather than specifying a single threshold (Table S6; see the Global soil suitability mapping section and Table S8, for more information on how our suitability classifications incorporated these uncertainty ranges).

Recovery product characteristics relevant to soil suitability. Each recovery product has distinctive characteristics that may affect its suitability relative to one or more soil parameters. For wastewater, we assumed soil suitability characteristics are similar for both aerobically and anaerobically treated waters. The nutrients in treated wastewater tend to be present as soluble ions, making them highly mobile and potentially prone to retention issues (i.e., losses). Nutrients may leach from coarse-textured soils, while phosphorus fixation may occur in weathered soils. In either case, limited crop nutrient utilization may reduce the product's efficacy, and achieving desired crop yields may require greater inputs than in soils without these retention and availability issues.

During storage of source-separated urine, spontaneous urea hydrolysis results in an alkaline product that may compromise soils with high pH or help to increase pH in acidic soils. Nutrients are more concentrated in urine than in wastewater, but these products are similar in that nutrients are highly mobile, creating potential retention issues in certain soils. Additionally, both urine and wastewater can contain high concentrations of sodium ions[5], which may exacerbate conditions in sodic soils and further limit nutrient retention, especially in arid climates with limited water for sodium leaching.

In compost and digested sludge, at least some nutrients are bound in organic compounds that require mineralization to become available to crops. Accordingly, nutrients are less mobile and potentially less prone to issues of low retention, but they are also less immediately available for crop uptake. Nutrient benefits from these products may not become apparent until future growing seasons. However, beyond increased nutrient supplies, the organic matter contained in compost and sludge represents a valuable non-crop nutrient contribution to soils, potentially improving structure, erosion resistance, and nutrient and water absorption and retention[198,209].

Among crystalline products, ammonium sulfate is an acidic compound that is unlikely to be detrimental to or may even benefit alkaline soils, but it may be detrimental for soils with low pH. It is highly soluble, making its nutrients highly mobile. In contrast, struvite has a high pH and is less water-soluble, suggesting it could act as a slow-release fertilizer. However, acidic soil conditions may cause struvite to dissolve more rapidly, increasing nutrient mobility[78]. Struvite may also benefit sodic soils, as its magnesium could displace sodium ions from the soil exchange complex.

Global soil suitability mapping. The relationships between soil parameters and product characteristics suggest conditions in which each recovery product may be detrimental or beneficial to agricultural soils, or where a given product's efficacy (i.e., ability to deliver nutrients to crops) may be diminished. Using global maps of soil parameters from the Harmonized World Soil Database[203], we applied the criteria defined for each parameter (Tables S6-S7) to assess the suitability of each recovery product based on its characteristics (Tables S4-S5). Given the available resolution of the global soil database, we generated global suitability maps having a resolution of 0.5 x 0.5 arcmin (approximately 1 km^2 at the equator). In all locations, we focused on values reported for the soil surface layer(s) (0-30 cm depth)[203]. Criteria that would classify a product as detrimental in a given location took the highest precedence (for example, a product that is detrimental relative to one soil parameter and beneficial relative to another was classified as detrimental). Beneficial characteristics were next, followed by characteristics related to limited efficacy. Within this final category, characteristics affecting general nutrient utilization took precedence over those specific to phosphorus fixation. If no product characteristics were classified as being detrimental, beneficial, or related to reduced efficacy in a given location, the product was classified as "acceptable" there.

The range provided for each soil criterion represents an uncertainty range (Table S6). If a location's parameter value relevant to a given product was within the uncertainty range, we characterized the product's suitability as being "potentially" affected. Parameter values beyond the uncertainty range suggested suitability was "likely" affected. For example, if soil pH is 8.0 (within the alkaline criterion's range of 7.2-8.5), an alkaline recovery product such as urine would be classified as "potentially detrimental" in that location. Alternatively, urine would be classified as "likely beneficial" if soil pH is 4.2 (beyond the acidic criterion's range of 4.5-5.5). This system resulted in nine product suitability classifications (listed from highest to lowest precedence): likely detrimental, potentially detrimental, likely beneficial, potentially beneficial, likely limited nutrient effectiveness, potentially limited nutrient effectiveness, likely limited phosphorus availability, potentially limited phosphorus availability, acceptable (see Table S8 for further details). While this categorical classification system cannot adequately capture all local factors associated with each recovery product, we feel it provides a reasonable first estimate of potential suitability from a global viewpoint. Incorporating even the coarsest information regarding soil context could markedly improve global assessments concerning the contextual appropriateness of nutrient recovery strategies.

Nutrient recovery potential from newly-installed sanitation systems. To estimate the quantities of nutrients that could be recovered from sanitation systems in different forms, we began by using procedures from previous work[167,202] to generate spatially-resolved estimates of nutrient excretion, based on population density and country-level per capita protein and calorie intake (Section S2, Table S9). These procedures incorporated a Monte Carlo analysis with Latin Hypercube Sampling[133] (10,000 runs) to produce distributions of likely excretion rates. In each country, we extracted median, 5th percentile, and 95th percentile values from these distributions to represent expected, low, and high nutrient excretion scenarios, respectively.

To estimate nutrient recovery from excreted urine and feces entering sanitation systems that will need to be installed to achieve universal basic coverage (subsequently referred to as newly-installed sanitation), we assumed that each recovery product was generated under either combined stream processing or source-separated treatment (Section S3). Each option represents potential recovery from the given waste stream and assumes the process is engineered to optimize production of the given product (Table S10). Combined processes included aerobic (activated sludge) and anaerobic (upflow anaerobic sludge blanket) wastewater treatment, while source-separated urine was treated via closed storage. Crystalline products were generated from separated urine (as it contains 74-93% of total excreted nitrogen, 33-75% of phosphorus, and 53-93% of potassium[20,48]). Compost and sludge were produced from separated feces. Although nutrient recovery from separated feces is relatively low, high total recovery can still be achieved by capturing nutrients from both source-separated urine and fecal streams in parallel.

For all products, three recovery scenarios reflected expected, low, and high recovery efficiencies (based on the uncertainty bounds in Table S10). Combining these recovery efficiencies with estimated excretion rates, three overall scenarios for nutrient excretion and recovery were defined as follows: expected (median excretion rate in each country, expected recovery efficiency for each product), low (5th percentile excretion, minimum recovery), and high (95th percentile excretion, maximum recovery). Together, these scenarios produced a broad range (including worst and best cases) of potential nutrient recovery from newly-installed sanitation systems.

Co-location and soil suitability of nutrients recovered from newly-installed sanitation. In our analysis, co-location refers to the degree to which recoverable nutrients spatially align with agricultural nutrient requirements. We used procedures from previous work to estimate spatial distributions of agricultural nutrient demands based on harvested areas and fertilizer recommendations for 52 crops[202]. We then compared nutrient demand with potential recovery to estimate co-location, using procedures similar to those developed in previous work[167]. The nutrient quantities present in a given recovery product were compared with agricultural requirements in the same cell, and we calculated co-location as the fraction of the product that could be applied without exceeding nitrogen, phosphorus, or potassium demands.

We then evaluated the soil suitability of each co-located product. The co-located quantity in a given grid cell was assigned the suitability classification specified for that cell in the product's suitability map. It should be noted that results from this simplified spatial assessment should be taken as first-order estimates of product co-location and suitability. Recovery products could be transported beyond the grid cell in which they are generated, relocating nutrients to areas with better suitability characteristics and greater crop demands[202]. The scope of this global exercise excluded transport beyond the initial grid cell.

Finally, results were aggregated, globally and by country, to estimate the percentages of each recovery product co-located with crop demands and falling within each suitability category.

To provide quantitative estimates comparable across countries, we report the nutrient mass from each product in each suitability category, normalized to each country's total cropland area (Tables S11-S14).

Results and Discussion

Understanding local soil conditions is critical in fully characterizing the value proposition associated with different forms of nutrient recovery. To examine the soil suitability of seven recovery products (reclaimed wastewater, source-separated urine, digested sludge, compost, ammonium sulfate, ammonium struvite, and potassium struvite), we identified several soil parameters that may be affected by a given product or impact the product's ability to meet crop nutrient demands. We focused on parameters relevant to crop production, as agricultural application is a straightforward and commonly promoted use of recovered nutrients[29,167]. The key parameters we considered were pH, sodicity, soil cation exchange capacity (CEC), clay content, and clay fraction CEC (see Section S1 for descriptions of why specific parameters were included and Tables S6-S7 for a summary of suitability criteria). Soil organic carbon (SOC) was not directly included, due to a lack of general guidelines defining desirable levels[210–212]. However, especially where SOC has been depleted[213], organic-rich recovery products (e.g., compost, sludge) are likely to benefit agricultural use of soils by reducing erosion, elevating soil organic matter content, and increasing nutrient and water retention[198,209] (see Section S1 for additional details on SOC). To accommodate a global scope, employ existing data, and enable transparent, communicable findings, we focused on a relatively limited set of parameters critical to agricultural soil conditions around the world, acknowledging the selected parameters are not always mutually independent or fully representative of relevant contextual factors.

Global soil suitability mapping of recovery products. Interactions between soil parameters and recovery product characteristics can determine where a product may be most suitable relative to local soil quality, or where it may have limited efficacy (Figure 4.1). Soil pH tends to play the largest role in determining suitable locations for several products. Alkaline products (urine and struvite) are classified as “detrimental” in regions with high soil pH since application (and dissolution in the case of struvite) would exacerbate growth inhibition due to alkaline soil conditions (“potentially detrimental” indicates local pH is within the uncertainty range defined in Tables S6, while “likely detrimental” denotes local pH beyond the uncertainty range; see Methods and Table S8 for further description of this nomenclature). These areas cover large swaths of several continents, often corresponding with arid environments (e.g., the Sahara, the Gobi). Formation of carbonate salts tends to contribute to desert regions’ alkaline conditions. Conversely, alkaline products may benefit acidic soils in large areas of North and South America, central Africa, northern Eurasia, and southeast Asia. However, some of these locations represent large, unmanaged forests (e.g., the Amazon, the Congo), suggesting agricultural application may be less likely. Locations of potential benefit and detriment associated with acidic ammonium sulfate are essentially the reverse of those for alkaline products. As each crystalline product tends to focus on recovery of one or two nutrients and is either acidic or alkaline, recovering and applying multiple products may buffer against pH changes while supplying multiple nutrients.

High sodicity affects relatively small areas mostly in South America and central Asia (Figure S2). Products with high sodium levels (urine, wastewater) may exacerbate sodium toxicity in these soils. Struvite could prove beneficial in these areas, as its magnesium may displace sodium from the soil exchange complex for potential leaching out of the crop root zone. However, many sodic soils are also alkaline. In our analysis, detrimental characteristics of a product take precedence over its benefits if both are locally relevant. Therefore, struvite is classified as detrimental in these areas (Figure 4.1). For other products, sodic soils may limit nutrient retention and crop utilization, due to sodium saturation of the exchange complex.

Much larger areas of the world are susceptible to nutrient retention issues associated with low clay content or soil CEC. In particular, these conditions tend to drive the soil suitability of wastewater in many locations (Figure 4.1). For wastewater and other products with highly mobile nutrients (wastewater, urine, ammonium sulfate), low clay content or soil CEC may increase nutrient transport away from crops, limiting possible yield improvements. We did not classify struvite as having highly mobile nutrients. However, under acidic conditions, struvite may dissolve rapidly, increasing its nutrients' mobility[78]. In locations where nutrient retention may be a concern, clay content and soil CEC are both low (e.g., parts of Australia, southern Africa, and Russia; Figure S2). Similar geographic distributions of these parameters reflect the fact that the clay-sized fraction drives total soil CEC, due to the high specific surface area and reactivity of clay-sized minerals[214,215].

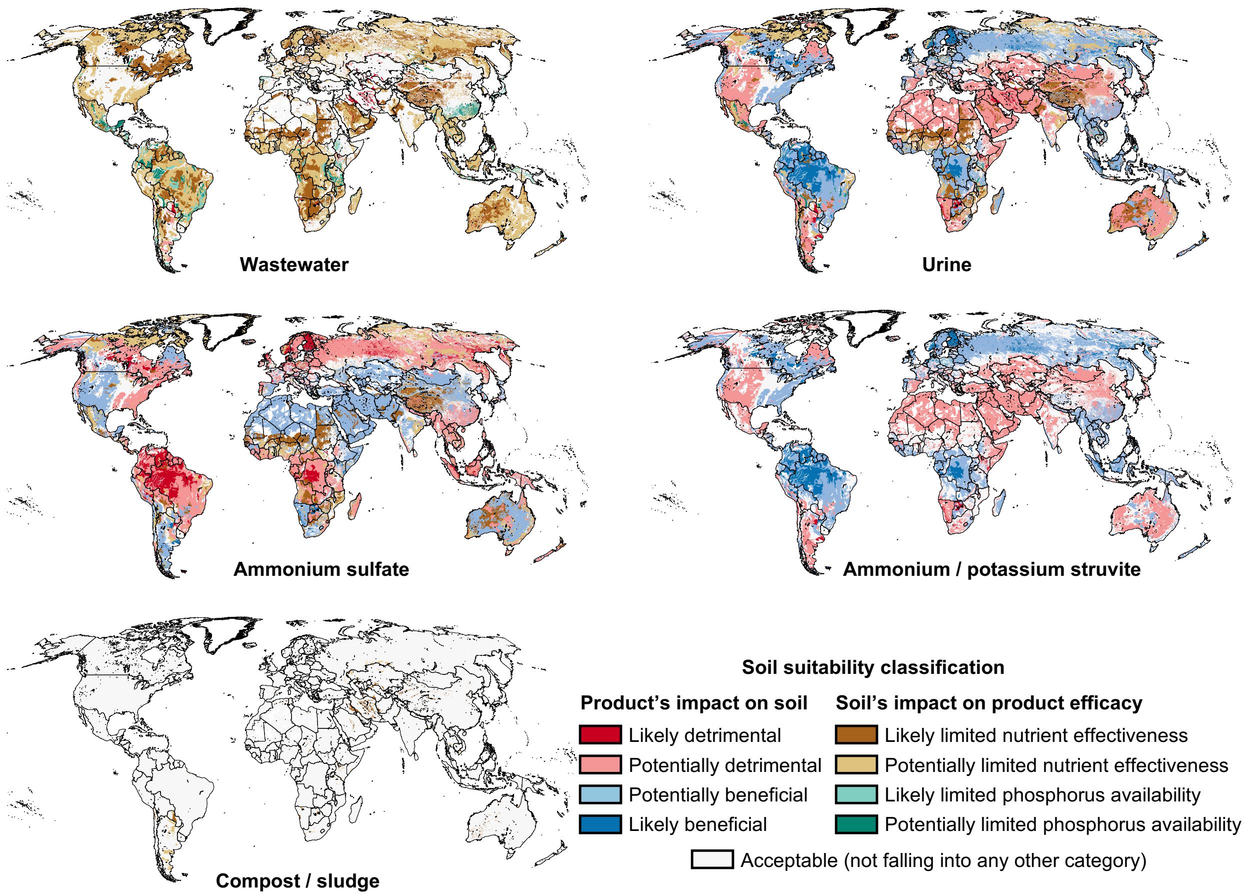

Figure 4.1. Global soil suitability maps for all recovery products. The coloring of each map shows where that product may impact soil conditions detrimentally (exacerbating one or more poor soil conditions for crop production, red) or beneficially (improving poor soil conditions, blue), and/or where soil conditions may limit a product's fertilizer efficacy (due to the soil's low nutrient retention, brown, or high phosphorus fixation capacity, teal). Locations not falling into any of these categories are classified as being acceptable (light gray). Shading also differentiates between "likely" impacts (where the relevant soil parameters fall beyond the uncertainty range in Table S6) and "potential" impacts (relevant parameters fall within the uncertainty range; see Methods and Table S8 for a more detailed description of this nomenclature). Ammonium struvite and potassium struvite are shown in one map, because these two products have similar characteristics. Compost and sludge also appear in one map. National administrative boundaries that provide the base of each map were taken from the Gridded Population of the World (Versions 3 and 4)[216,217].

Finally, phosphorus fixation (i.e., immobilization of phosphorus by irreversible adsorption and/or precipitation to metal cations, making the nutrient less accessible to crops) most often impacts highly weathered soils in tropical areas rich in aluminum and iron oxides (e.g., ferralsols, covering 7-8% of global ice-free land area)[214]. Low clay CEC can serve as a proxy for high weathering[214], with particularly low values in Brazil and central Africa (Figure S2). However, many of these locations were already prone to general nutrient retention issues, which took precedence (Figure 4.1). Given that total soil CEC is predominantly derived from the clay-sized fraction, a low clay CEC often aligns with low soil CEC. On our maps, areas with phosphorus fixation issues are especially uncommon for urine, as highly weathered soils – generally situated in the tropics – also tend to be acidic. Urine was already classified as beneficial in these locations.

Potential nutrient recovery from newly-installed sanitation systems. Global soil suitability mapping of recovery products represents an important category of information that has been lacking. However, these findings must be combined with several other factors (e.g., nutrient recovery potential, agricultural demands, social acceptability, economic viability) to more fully evaluate the locality-specific implications and appropriateness of nutrient recovery processes. Below, we illustrate such a combined analysis, although we do not consider all potentially important factors. We keep our scope focused on global nutrient cycles and local agricultural reuse of recovery products, acknowledging that other considerations will play into local sanitation decisions.

Achieving universal sanitation access is a Sustainable Development Goal (SDG)[12,69], and we focus this analysis on populations currently lacking basic sanitation to explore the opportunities associated with recovering nutrients from systems that will need to be newly installed to meet this target (referred to as "newly-installed sanitation"). Many populations without sanitation access face simultaneous challenges of resource access and economic security, suggesting resource recovery may generate multidimensional benefits[167]. For each recovery product, we leverage existing literature and previously-developed methods[167,202] to generate quantitative, spatially-resolved estimates of nutrient recovery potential from newly-installed sanitation (i.e., the nutrient quantity recoverable in a given product at a given location, based on population density, basic sanitation coverage, per capita nutrient consumption and excretion, and the product's potential recovery efficiency). We account for uncertainty around nutrient excretion and recovery by employing three scenarios (expected, low, and high; see Methods for full scenario description). We then estimate the degree to which nutrients present in recovery products are co-located (i.e., in the same grid cell) with local crop nutrient demands. Combining these results with our soil suitability maps, we generate quantitative estimates of nutrient recovery, co-location, and soil suitability for 158 countries with sufficient data (Figure 4.2, Tables S11-S14; see Sections S2-S3 for further details).

Across countries, the relative nutrient quantities recoverable in various products follow similar patterns (Figure 4.2). Anaerobically-treated wastewater could provide the largest nutrient quantities of any single product (in part because a combined waste stream contains nutrients from urine and feces). Under anaerobic conditions, microbial growth and nutrient uptake are lower than in aerobic conditions[5], allowing more nutrients to remain in the effluent. Anaerobic treatment may be particularly applicable in contexts with high organic loading[5] (e.g., fecal sludge from latrines). Regardless of treatment approach, adequate pathogen reduction is needed for safe wastewater reuse. Irrigation with partially treated or untreated wastewater can increase risks for diarrheal disease and helminth infections, especially among agricultural workers[218].

Products derived from source-separated urine can also capture substantial nutrient quantities. Urine itself can act as a liquid fertilizer after storage (especially when undiluted, urine's high pH and intrinsic ammonia content can reduce pathogen levels[61,219]), while crystalline products (ammonium sulfate, ammonium struvite, potassium struvite) recover nutrients in concentrated forms that may be easier to transport to more distant cropland[202]. Conditions under which crystalline products are generated determine bacterial inactivation and product safety[220]. Although no single crystalline product contains all three nutrients, a sequential configuration can generate multiple products (e.g., potassium struvite precipitation from fresh urine, followed by ammonium struvite precipitation, and finally stripping, absorption, and evaporation to recover remaining nitrogen as ammonium sulfate). Products derived from feces (compost, sludge) offer lower recovery levels, because urine contains most excreted nutrients (74-93% of nitrogen, 33-75% of phosphorus, 53-93% of potassium)[20,48]. However, if nutrients are captured in multiple products derived from both urine and feces, total recovery rates in these systems may be similar to those in anaerobic wastewater treatment.

Comparing across countries, total recoverable nutrient quantities from newly-installed sanitation depend upon factors including existing sanitation coverage, population density, and dietary intake (Figure 4.2; quantities are normalized relative to total cropland area for comparison). As illustrations, we discuss two countries with relatively high recovery potentials. India's recovery potentials are associated with the country's relatively low sanitation coverage and high population density (more people need newly-installed systems, and more nutrients are excreted per unit area). In Uganda, basic sanitation access is particularly low, while a rapidly rising population will increase population density in the future. However, Uganda's normalized recovery potentials are lower than India's. Potential explanations include Uganda's large cropland area (distributing nutrients across more of the country) and low nutrient excretion (due to disparities in protein intake[53], we estimate median nitrogen excretion in Uganda at 7.9 g $N \cdot cap^{-1} \cdot d^{-1}$, compared with 9.1 g $N \cdot cap^{-1} \cdot d^{-1}$ in India). Nevertheless, nutrient recovery in Uganda could greatly increase access to agricultural inputs, as the country's current use of inorganic inputs per hectare of cropland is <2% of the global average[53].

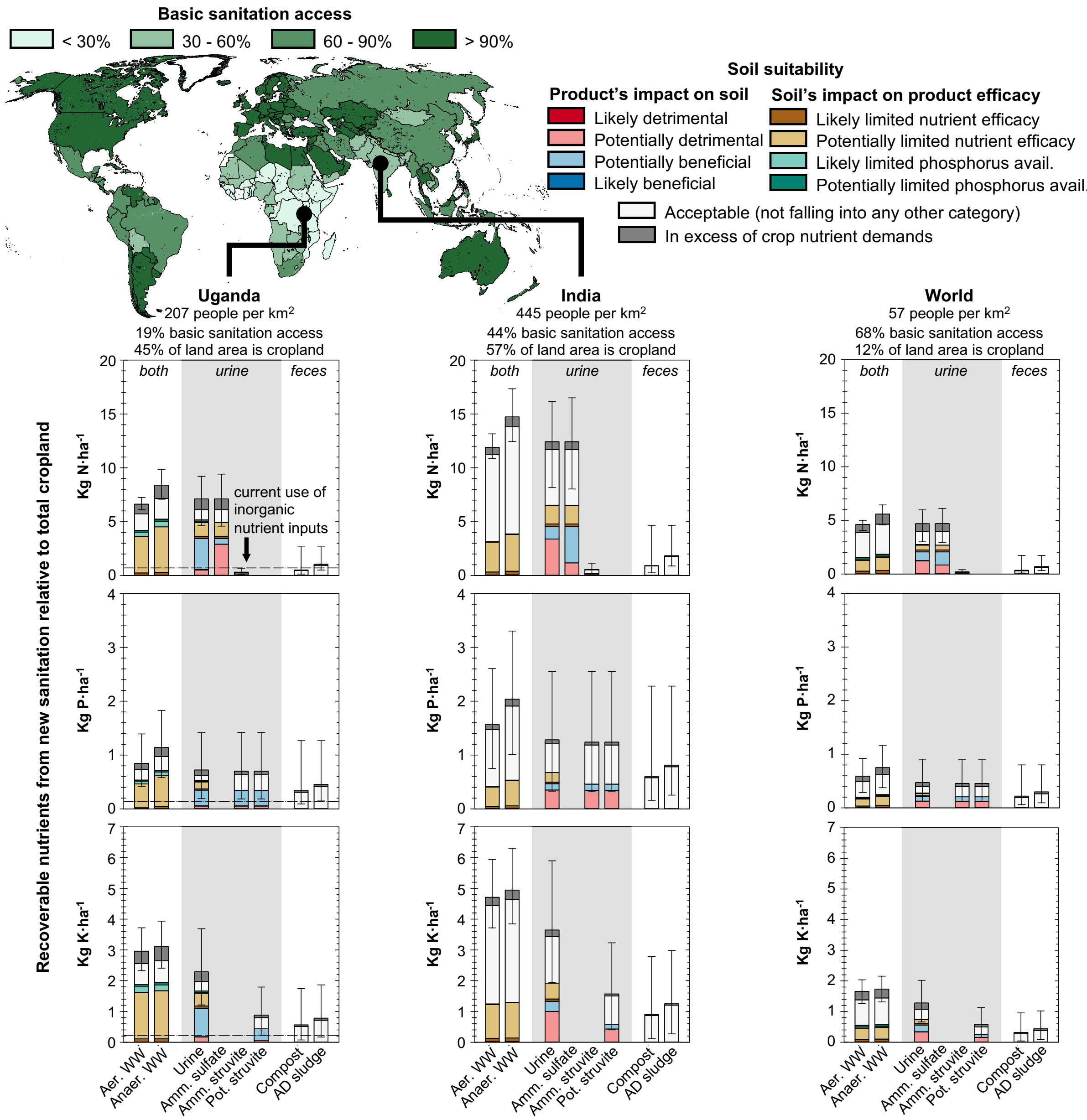

Figure 4.2. Nutrient recovery quantities and suitability from newly-installed sanitation systems. The world map shows each country's level of basic sanitation access in 2015. Bar graphs show estimated quantities of nutrients (nitrogen, phosphorus, potassium) that could be recovered if systems installed to achieve universal sanitation coverage are optimized to generate a given recovery product from a given waste stream, globally and in two illustrative countries (values from all countries and scenarios are in Tables S11-S14). Recovery products are grouped based on assumed source: combined waste streams (aerobically or anaerobically treated wastewater), source-separated urine (all crystalline products), or feces (compost, sludge). Each bar shows total recovery potential of that product from the assumed waste stream (normalized relative to total cropland area) in the expected excretion and recovery scenario, with error bars showing recovery potential in low and high scenarios. Missing bars indicate the given nutrient is not present within that product (e.g., ammonium struvite contains no potassium). Dark gray shading within each bar shows the fraction of the total recovered product in excess of crop nutrient demands within the same grid cell. For the remaining co-located fraction (not in excess), coloring indicates the suitability of that product relative to soil conditions in the same cell (using the same color scheme as Figure 4.1). The graphs for Uganda also show current levels of inorganic nutrient application (all from imported fertilizers)[53], because potential nutrient recovery exceeds this level. Average inorganic nutrient use levels in India (99 kg N·ha^{-1}, 17 kg P·ha^{-1}, 12 kg K·ha^{-1}) and the world (69 kg N·ha^{-1}, 13 kg P·ha^{-1}, 20 kg K·ha^{-1})[53] are higher than recoverable nutrient quantities. Population density[53], basic sanitation coverage[69], and cropland area[53] are also noted for each illustrative country and the world. National administrative boundaries that provide the base of the map were taken from the Gridded Population of the World (Versions 3 and 4)[216,217].

Recovery product suitability and decision-making. Overall, our results provide several layers of information potentially useful for local, regional, or global decision-makers. They offer estimates of nutrient recovery and co-location with local crop demands, and they suggest how these data can interact with information on the soil suitability of various recovery products to provide guidance for local or national sanitation strategies. For example, while Uganda could capture considerable quantities of human-derived nitrogen by recovering ammonium sulfate, applying this product to acidic soils common in Uganda may adversely affect crop production (Figure 4.2; although, application in relatively small quantities may provide much-needed nitrogen without impacting soil pH too severely). In contrast, focusing on recovery of ammonium struvite may provide valuable opportunities to alleviate acidic soil conditions with an alkaline product while providing nitrogen and phosphorus. In other places (India, for example), decision-makers who take these considerations into account may reach different conclusions (Figure 4.2), underscoring the importance of accounting for local soil context when assessing nutrient recovery alternatives.

However, global soil databases may not provide the best information to guide local, national, or regional decisions. The accuracy of global soil maps is likely to be less than that of continental or regional maps, and coarse-resolution maps may have limited usefulness for localized spatial planning[221]. For our analysis, we used global data (the Harmonized World Soil Database[203]) because the study's main proposition – that considering soil context can help inform sanitation and nutrient recovery strategies – is of global consequence. Nevertheless, to illustrate the importance of using contextually appropriate data, we repeated our analysis for Africa using a finer-resolution continental dataset (Africa Soil Information Service[221]). While the soil suitability of recovery products appeared similar across some parts of sub-Saharan Africa (Figure S3), other locations (e.g., Uganda) revealed considerable disparities between global and continental results (Figure 4.3).

For example, the global analysis suggested that up to 42% of the struvite recoverable from newly-installed sanitation systems in Uganda may have beneficial agricultural impacts given local soil conditions (e.g., moderation of acidic pH), whereas the repeated analysis based on the continental dataset estimated only up to 12% of recoverable struvite may beneficially impact local recipient soils (Figure 4.3; while we focused on local recycling as a logical first step in improving nutrient access, these percentages could be increased with transportation beyond the grid cell). In most locations throughout the country, product suitability classifications changed to "acceptable" (i.e., not meeting the criteria for any other category), likely because soil parameter values (e.g., pH, soil CEC) in the continental dataset vary to a lesser degree than in the global dataset. Beyond the "acceptable" category, the remaining quantity of each product tends to follow trends similar to those seen in the global analysis (e.g., in both analyses, most struvite not classified as "acceptable" falls into the "potentially beneficial" category). Despite considerable disparities between the analyses at these two scales, this comparison suggests that the global analysis may still provide useful information regarding general country-level soil suitability trends. It could serve as a mechanism to identify locations for more focused study with localized information.

Regarding the general issue of appropriate map scale and resolution, we would typically expect finer-resolution soil maps to be more accurate. More localized maps, therefore, might provide better information for local decision-makers, and the continental data may be more appropriate than the global database for countries in sub-Saharan Africa. Reliability may also depend on how maps are constructed (e.g., using soil profiles or remote sensing) and mapping focus (e.g., soil property or classification). In sub-Saharan Africa, some older country-level maps may be less accurate than newer continental datasets. Colonial-era maps used soil profiles and classification systems that are now outdated[222,223]. Deriving specific soil properties from these classifications entails some uncertainty, and conditions may have changed in the decades since the maps were developed. Overall, then, decision-makers should be aware of uncertainties associated with soil data and the benefits associated with developing soil maps. Enlisting soil and agricultural science experts to help navigate discrepancies in datasets can increase the likelihood that appropriate information is being used when developing or assessing strategies for sanitation and resource recovery.

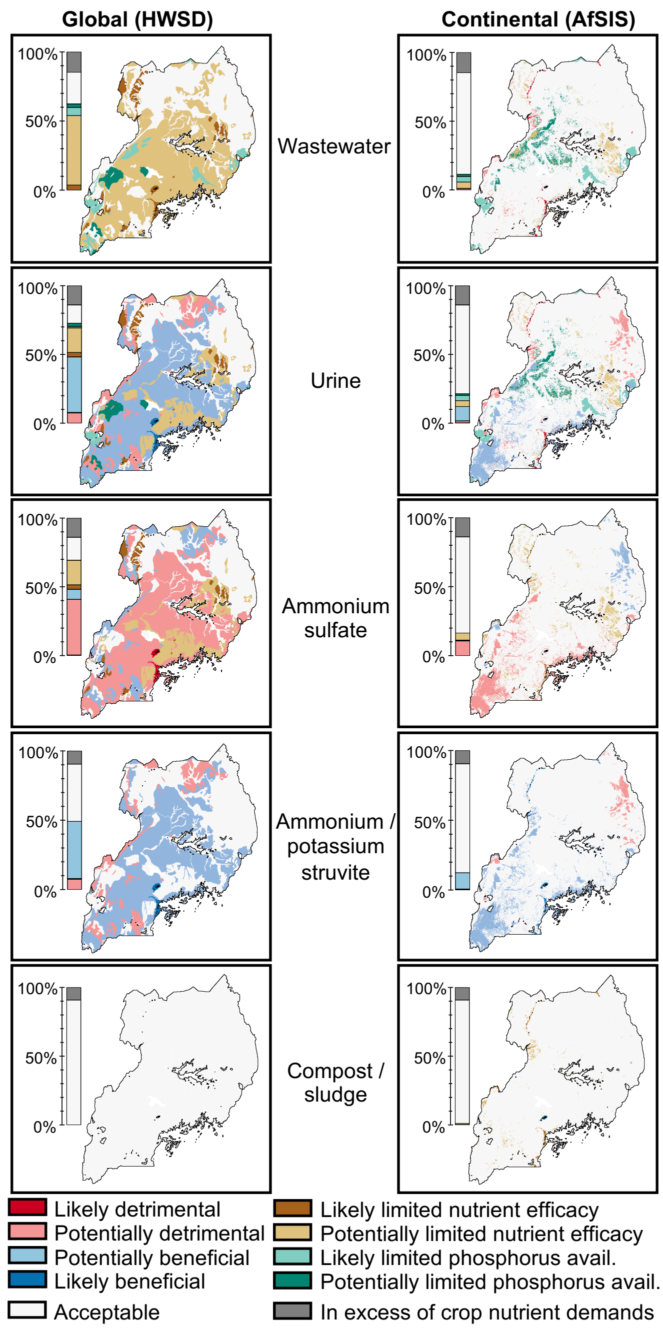

Figure 4.3. Comparing soil suitability findings in Uganda using two datasets. Maps on the left show soil suitability of each recovery product, using the soil dataset employed in our global analysis (Harmonized World Soil Database, HWSD[203]). On the right are results from the same procedure when using a continental soil dataset (African Soil Information Service, AfSIS[221]). The bar graph inset in each map shows the fractions of that product (recoverable from new sanitation systems) that fall into each suitability category. While results generated from these two datasets are similar for some locations in sub-Saharan Africa (Figure S3), Uganda represents a case with considerable discrepancies, highlighting the importance of using appropriate and accurate soil maps when making decisions about nutrient recovery from sanitation. National administrative boundaries that provide the base of each map were taken from the Gridded Population of the World (Version 4)[217].

Implications. This study illustrates the importance of explicitly considering soil context when developing, assessing, and making decisions regarding locally appropriate sanitation, nutrient recovery, and agricultural reuse systems. Certain nutrient recovery techniques and products may appear economically or logistically feasible in a given setting, but decision-making processes should also factor in whether application of that product might help or hinder crop production through interactions with local soil conditions. Alternatively, stakeholders may explore product export to locations in which soil contexts are more favorable. The global analysis presented here has numerous limitations, primarily related to the accuracy of available global datasets and the simplifications necessary to apply a generalized analysis across a wide range of contextual conditions. We acknowledge these uncertainties through our soil suitability classifications ("likely" versus "potential" effects) and the use of multiple nutrient excretion and recovery scenarios. As such, our results represent first-order estimates of potential nutrient recovery and soil suitability, which can improve context-specific assessment at a global scale and reveal general trends and locations for further, more focused investigation.

Recycling nutrients from human sanitation may be especially valuable for smallholder farmers, many of whom live in lower-income nations where considerable gaps in sanitation access persist and agricultural yields are often constrained by nutrient and water limitations[69,144,194]. These conditions highlight the synergistic potential of resource recovery systems in addressing multiple SDGs simultaneously[167]. However, along with factors such as transport distance[76,202] and the inputs required by recovery processes (e.g., energy, chemicals)[157,207,208], local soil conditions are critical to the value proposition of nutrient recovery. Each of these considerations may have locality-specific impacts on the financial viability of nutrient recovery and agricultural reuse. In some contexts, for example, high magnesium costs may discourage struvite precipitation[157], while long transport distances may constrain the utility of less concentrated recovery products[202]. Soil context may limit retention but also crop uptake of nutrients provided through certain recovery products, while other products may worsen conditions that hinder crop production, potentially reducing agricultural income. Alternatively, some products may provide benefits beyond their nutrient content, improving soil conditions and potentially increasing crop yields. As such, incorporating even the coarsest information regarding soil context could markedly improve global assessments concerning the contextual appropriateness of nutrient recovery strategies. Together, local experts, policy-makers, farmers, utilities, and other stakeholders can incorporate this information into decision-making processes that account for multiple factors to develop and implement appropriate and sustainable sanitation solutions.

CHAPTER 5: RESOURCE RECOVERY FROM SANITATION TO ENHANCE ECOSYSTEM SERVICES[d]

Introduction

Ensuring universal access to safe and equitable sanitation represents a critical aspect of the United Nations' Sustainable Development Goals (SDGs) with potentially far-reaching implications. At present, global shortfalls are considerable, with an estimated 4.5 billion people lacking safely-managed sanitation access[224]. The potential to recover resources, including nutrients, organic matter, and water, has emerged as a possible avenue to increase the real and perceived value of sanitation systems beyond their primary function of mitigating environmental and human health risks. This added benefit holds particular importance as it can increase adoption and financial viability of safe sanitation, particularly in resource-limited settings where coverage levels are low[15,167].

Society may derive further value from these recovered resources through ecosystem services (ES)[87]. ES such as food and water provisioning, nutrient cycling, and climate regulation can contribute to several SDGs, including those related to reducing hunger, sustaining aquatic and terrestrial life, ensuring clean water, developing sustainable cities, and promoting climate action[88,89]. Sanitation is typically seen as being passively dependent on or improved by ES (e.g., wastewater treatment in natural wetlands, pollutant assimilation in rivers, lakes, estuaries, or oceans)[87,88,225], with less thought given to the more proactive role it might have. Indeed, a preliminary bibliometric analysis shows limited crossover between sanitation/resource recovery and ES, despite rising publication rates in each field (Figure 5.1; Supplementary Results; Supplementary Table 1). Existing literature overlap often emphasizes how wastewater treatment,

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water reuse, and wetlands can connect to water purification services. However, a few publications suggest integrated design paradigms can expand the engineering design space to harmonize technological and natural processes and develop synergistic approaches to meet societal and ecosystem needs[90,91]. Beyond nature's contributions to people[226], recovered resources represent materials society can contribute back to ecosystems, thereby supporting a positive cycle of reciprocal benefits (e.g., by enhancing services such as erosion control and food provisioning through organic matter and nutrient application). However, sanitation or water management frameworks explicitly incorporating ES beyond water quality improvement[92] remain rare.

Resources recovered from sanitation systems can offset global pressure on nutrient, energy, and water systems already straining planetary boundaries[64]. They also promise to increase resource availability in low-income settings[29,167]. However, implementation of innovative sanitation approaches has been limited by inertia related to existing infrastructure and planning organizations, issues of public acceptance and perceived risks, and economic analyses that neglect ecological benefits. Frameworks that integrate the value of enhancing ES represent a more holistic view of sanitation – one that may reveal greater opportunities for resource recovery. Furthermore, funding mechanisms such as payments for ecosystem services (PES) may enable sanitation to improve, restore, or sustain ES. Globally, public and private entities employ over 550 active PES programs to compensate existing conservation efforts, providing a total of $36-42 billion in annual transactions[227]. Particularly in settings with considerable ecological assets but limited economic means, similar types of support for sanitation could amplify sustainable development efforts in ways that go beyond its current real and perceived societal role.

Accordingly, the objectives of this article are (i) to characterize the pathways through which resource recovery from sanitation and ES can generate reciprocal benefits, and (ii) to propose a course via which the international community may begin to assess the viability of leveraging these relationships to enhance sustainable development. Here, we bring together literatures on resource recovery from sanitation and ES to place these fields – generally regarded as disparate in practice – in creative dialogue with one another. We develop a framework of potential pathways through which recovered resources and ES can intersect and generate societal value. Moving further, we begin to identify and examine multiple factors that may affect the contextual feasibility of leveraging these connections. We integrate an analysis of co-location between recoverable resources and land cover types to investigate spatial viability, and provide additional discussion of sanitation technology selection and financing mechanisms. Overall, this work advances knowledge of how research and policy efforts to link sanitation and ES will enhance sustainable development goals and circular economies relevant to both fields.

A conceptual framework linking recoverable resources and ecosystem services

As most literature does not directly frame the impacts of recovered resources in the context of ecosystem services (Figure 5.1; Supplementary Results), sanitation researchers may miss opportunities to engage the ES community and explore the ecological value of resource recovery. We begin to address this gap by drawing on literature from both fields to develop a conceptual framework that explicitly connects resources with supporting, regulating, and provisioning ES (as defined by the Millennium Ecosystem Assessment [MEA] and related studies[87,88,225,228–230]), identifying service "pathways" that often include multiple ES and lead finally to “final services” of direct societal value (Figures 5.2-5.3). A pathway-focused approach conceptualizes sanitation facilities and ES as integrated components of sustainable cycles directing resources toward beneficial uses and away from unintended environmental consequences. Such an integrated design paradigm can enable exploration of mutually-beneficial interactions between engineered and natural systems, and it respects critiques arguing that some ES classification schemes double-count services that only support others without providing direct human benefits[228,229]. We follow the MEA’s general categories (rather than including services only applicable in certain contexts) to keep the framework broadly relevant and do not include cultural services as an isolated category because recent work describes culture – highly dependent on context[228] – as permeating through all of nature's contributions to society[226]. While endeavoring to reflect the complexity of potential relationships, we do not enumerate all possibilities. Rather, we present a framework suggesting opportunities for linkage in diverse contexts. Below, we begin by introducing three general categories of recoverable resources and examples of service pathways relevant to each (Figure 5.2), after which we transition to a more comprehensive discussion of the pathways delineated in our framework (Figure 5.3).

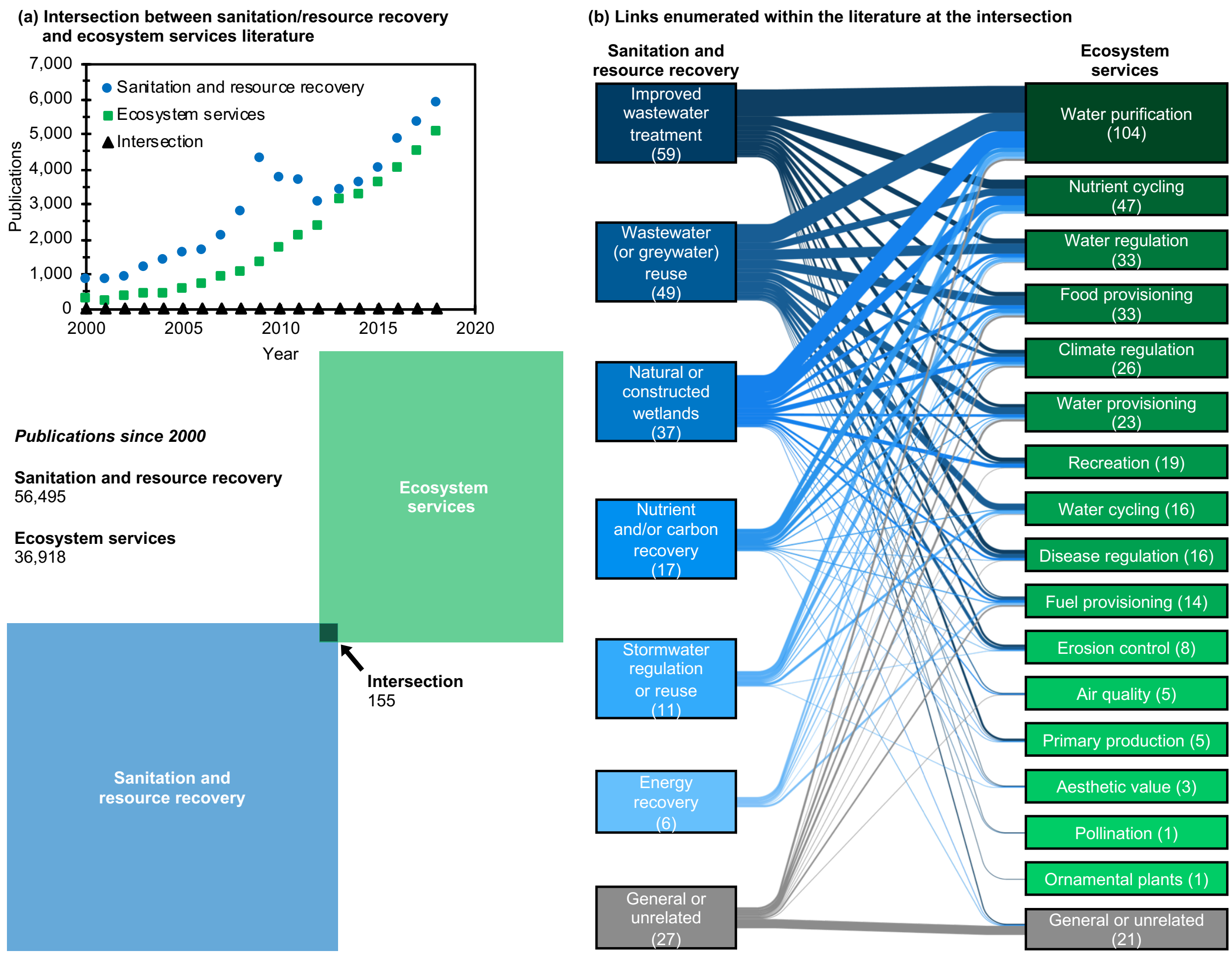


Figure 5.1. Literature review of the intersection between sanitation/resource recovery and ecosystem services. Crossover between all works published in both fields since 2000 (colored boxes, representing numbers of publications as of February 23, 2019) and annual publications from 2000 to 2018 (scatter plot) is shown on the left (a). The diagram on the right (b) presents the linkages enumerated in the 155 publications identified at the intersection between the two fields. Blue boxes represent different types of sanitation and recovery approaches, while green boxes represent ecosystem services (numbers in parentheses show how many of the 155 publications mention each approach or service; a single study may consider multiple alternatives). The width of each flow running from a sanitation/resource recovery approach to an ecosystem service signifies the number of publications mentioning a specific connection through which the sanitation/resource recovery alternative can enhance that service. Publications often include multiple connections. Some publications identified through the search were only tangentially related to sanitation and/or ecosystem services, or only mentioned these topics in general terms. Those cases are included in the gray boxes at the bottom of the diagram. Please refer to the Supplementary Results and Supplementary Table 1 for additional information regarding this literature analysis and its results.

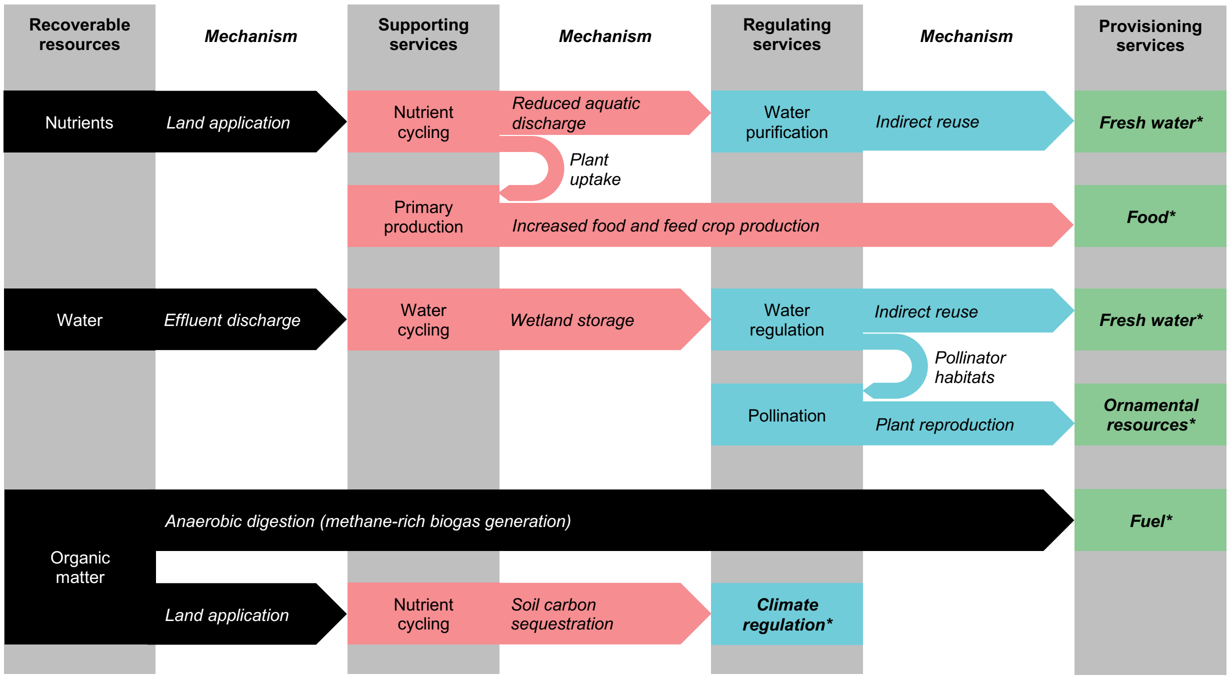


Figure 5.2. Examples of service pathways through which recoverable resources and ecosystem services can generate direct societal value. For each resource category (nutrients, water, organic matter), we define pathways through supporting, regulating, and/or provisioning services. Arrows linking one step to the next indicate mechanisms through which inputs can enhance subsequent services. Pathways may include multiple steps in the same service category (e.g., nutrient cycling, primary production) or may bypass certain categories (e.g., organic matter directly linked with fuel provisioning). The pathways shown here can also act as examples for interpreting Figure 5.3, where relationships between inputs, mechanisms, and enhanced services are presented in matrix form. For a given input listed in the left-most column of Figure 5.3, mechanisms for enhancing services are identified by moving horizontally to the right. The corresponding enhanced service (listed in the top row of the matrix) can then be found in the left-most column of the same or another matrix, thereby acting as the input for a subsequent step in pathway(s) leading toward one or more final services. *Final services (i.e., those likely to be of direct value to human populations) are shown in bold/italics.

Bodily waste managed in sanitation systems contains nutrients (nitrogen, phosphorus, potassium) and organic matter that can be recovered in various forms, and water can also be captured and treated for reuse[15,174,231]. Nutrient recovery processes may generate liquid or solid products (e.g., nutrient-rich water, sludge, concentrated products such as struvite[157] or ammonium sulfate[181]), and each product's contextual appropriateness depends on local factors including transport requirements, soil context, resource access, markets, and local behaviors[157,167,202]. Land application of these products can advance pathways benefiting agricultural systems. With their reintroduction into nutrient cycles (a supporting service), these resources are available for crop uptake to enhance primary production (another supporting service), eventually improving food provisioning (a final service of direct societal value; Figure 5.2)[209,232]. This pathway could increase farmers' access to agricultural nutrients in resource-limited settings[29,167] and/or offset application of inorganic inputs dependent upon regionally-concentrated minerals (e.g., phosphate rock) and energy-intensive conversions (e.g., the Haber-Bosch process)[30,66]. Furthermore, this approach can reduce aquatic discharges that dislocate nutrients and disrupt natural biogeochemical cycling (particularly when treatment does not include enhanced nutrient removal[174]). Nutrient pollution of water bodies causes eutrophication and algal blooms, degrading water sources and supplies. This process is already causing considerable environmental and economic damage in locations such as the Gulf of Mexico[1,225]. Capturing and applying nutrients to soils could function as a mutually-beneficial alternative or supplement to technocentric nutrient removal processes[90], improving treatment, resource cycles, and food production while reducing environmental decline.

Organic matter can be a potential energy source, generated through processes such as direct combustion of solids or generation of methane, hydrogen gas, or electricity[15,231,233]. This possibility reflects a direct connection between a recoverable resource and fuel provisioning services (Figure 5.2). While the impacts of recovery vary considerably, the potential for energy from organic matter to increase household energy access tends to be lower than the potential of nutrient recovery to improve farmers' resource access[167]. A typical household in Uganda might meet approximately 10% of its cooking needs with a biogas stove (utilizing methane from anaerobic digestion), while bodily excreta from a United States household can offset <1% of that household's energy use[167]. Beyond energy generation, land application of recovered organic solids can support carbon cycling to replenish depleted soil carbon[213] or increase soil carbon sequestration[232]. This practice may be more valuable than energy recovery in at least some scenarios. Indeed, it presents another example of mutually-beneficial interactions between sanitation and ES, as it may simultaneously reduce aquatic pollution and atmospheric carbon emissions while improving soil characteristics and crop yields[90,209,234].

Water recovery from sanitation is valuable[15,231] but highly dependent upon technology and system configuration. For example, pit latrines and waterless toilets use essentially no water, while one person may use up to 66 $L \cdot d^{-1}$ in flush toilets[235]. Existing sewer systems often combine greywater (i.e., water from bathing, washing, or other uses not associated with fecal contamination)[6] with toilet-flushing water, while other configurations may treat greywater separately (reducing treatment requirements before reuse or discharge[6]). Our general framework accommodates various sanitation configurations that may or may not separate greywater. In either case, discharge of appropriately-treated effluent into wetlands can help to ensure a reliable supply of water and prevent degradation of these diverse ecosystems. While enhancing water cycling, storage, and flow regulation functions, wetlands may also offer opportunities for natural contaminant uptake to supplement engineered treatment processes, contributing to improved quality of freshwater resources. Well-functioning wetlands can also provide healthy habitats for pollinators, which are instrumental in the plant reproduction processes required for many provisioning services (e.g., ornamental resources; Figure 5.2).

Now that we have provided several examples of service pathways that integrate recoverable resources and ES (Figure 5.2), we transition to a broader discussion of additional connections and pathways that can be drawn. Our full framework (Figure 5.3) presents these relationships as matrices directed toward supporting, regulating, and provisioning services. Beginning with a given resource (or intermediate ES) and moving horizontally to the right, we identify mechanisms through which the input may enhance one or more ES listed along the top. The enhanced ES can then act as an input in the same or another matrix, contributing to subsequent steps leading toward final services of direct societal value. We constructed the preceding examples using this process, which can generate numerous service pathways relevant in various contexts.

Recoverable resources and supporting services. As the name implies, supporting services may not immediately benefit humans, but they often support other services and constitute the first step on pathways toward societal value[87,228]. For example, effective nutrient and water cycling advance multiple pathways toward food production. In many places (particularly sub-Saharan Africa), current gaps in crop yields primarily result from limited availability of nutrients and/or irrigation water[144]. In Uganda, some farmer yields represent only 10% of research plot yields[236]. Meanwhile, anthropogenic alterations to water cycles have reduced river levels and depleted groundwater reserves. The Colorado River, for example, no longer regularly reaches the ocean, and its delta's wildlife habitats, biodiversity, and ecological functioning have declined accordingly[237]. Restoring water and nutrient cycles can improve degraded ecosystems, acting as a first step on many service pathways.

Recovered resources can also enhance soil formation and primary production, two key supporting services in many ecosystems[87]. Land application of organic solids can improve soil formation processes by building up stores of organic matter, regulating pH, and providing structural materials. These effects may be particularly beneficial in degraded landscapes (e.g., mine tailings disposal sites) to begin revegetation processes[238]. Finally, nutrients, water, and organic matter amplify primary production (i.e., synthesis of organic material from energy and nutrients[87]) by providing raw materials (mediated through nutrient and water cycles) and contributing to the formation of a favorable growth medium (soil). Primary production then supports a host of services reliant on plant growth (e.g., food and fiber provisioning).

Recoverable resources and regulating services. The impacts of recoverable resources on various regulating services are often mediated by supporting services. One exception to this pattern involves erosion control. Organic matter reduces erosion potential by directly improving soil's overall structure, stability, and water retention capacity[209,239]. In contrast, as examples of indirect connections mediated by supporting services, enhanced primary production of plants spurs root growth to improve soil retention[87], while long-term soil formation processes can replenish eroded soils. The benefits of erosion control can create a positive reinforcing loop, in which improved stability enhances nutrient, water, and soil retention and vegetative growth, contributing to even greater soil stability[230]. In much of Africa, erosion has caused severe cropland degradation and nutrient losses, contributing to relatively stagnant agricultural productivity over several decades[29,230]. In certain contexts, the erosion control benefits of organic matter may be highly valuable and should not be overlooked in favor of energy generation. Indeed, erosion control can contribute to other regulating services including water regulation, water purification, pollination, air quality, and climate regulation by reducing runoff losses, stabilizing pollinator habitats, and improving land's carbon-holding capacity.

Other pathways to enhance water regulation or purification may use treated water to recharge aquifers or streams, diluting contaminant concentrations and offsetting surface and groundwater depletion[237,240,241]. Recoverable resources' contributions to improved soil formation and primary production can also support water regulation and purification through soil filtration and contaminant uptake by plants (e.g., in wetlands)[87,225,230]. Additionally, restoring natural biogeochemical cycles by recovering nutrients, organic matter, and polluted water (rather than discharging them into water bodies) can prevent eutrophication and other negative impacts, helping to maintain aquatic ecosystems' natural treatment capacity. Purification processes (e.g., soil filtration) can contribute to disease regulation by directly reducing pathogen levels, whereas water cycling and regulation can reduce standing water that serves as a breeding ground for disease vectors (e.g., mosquitoes)[242].

Finally, air quality and climate regulating services often relate to contaminant uptake and carbon storage provided by vegetation in forests and other ecosystems (mediated through primary production)[87,225,229]. Alternatively, innovative wastewater treatment can directly contribute to carbon capture through phototrophic processes such as microalgae cultivation, which can generate biofuel or bioproduct feedstocks[243].

Recoverable resources and provisioning services. Literature on resource recovery from sanitation focuses attention on potential provisioning of food, fuel, and water[6,15,29,167,231], although these benefits are not typically framed as ecosystem services, and other services often function as intermediaries or alternative avenues to realize final societal value. Freshwater provisioning, for example, can be enhanced through indirect reuse mediated by surface or groundwater recharge (water regulation) and natural purification processes (e.g., wetland treatment, soil filtration, nutrient cycling away from water bodies)[1,240,241]. Additionally, indirect reuse occurs (often without the conscious awareness of the general populace) when downstream populations abstract water from the same river into which upstream populations discharged waste. Climate regulation may also reduce the potential for water stress by mitigating changes in global water cycles[244]. Alternatively, after appropriate treatment, direct water reuse for potable or non-potable applications (e.g., irrigation) presents additional opportunities for reducing water stress and supporting agricultural services[6]. Greywater reuse (likely associated with lower treatment requirements and public resistance) may be particularly beneficial, and recovering greywater's thermal energy (which is often larger than organic matter's chemical energy) could provide opportunities to offset other energy sources[6,17].

Similarly, provisioning of plant-based fuels (e.g., firewood) can be enhanced through recovered resources' effects on soil formation, primary production, pollination, and climate regulation, while organic matter can be used to generate energy directly[15,167,231]. Direct energy recovery (e.g., through anaerobic digestion of organic matter) can lead to additional value by reducing deforestation, biofuel cropland requirements, and fossil fuel use, thereby enhancing erosion control, air quality, climate regulation, food, and other services. However, as discussed previously, cycling organic matter toward final services such as food provisioning and climate regulation (through land application) may represent greater value than use as an energy source.

Remaining provisioning services (food, fiber, ornamental resources, biochemicals) relate to products generated through agriculture or acquired from natural environments (biofuels also fall into this category). Generally, recovered resources can increase production of these services through pathways that include soil formation, primary production, pollination, irrigation, and climate regulation (mitigating changes in temperature, precipitation, or growing season). Regarding food provisioning in particular, the potential food security benefits of nutrient recovery are a major focus of the literature[29,167], especially given that future production must accommodate growing and increasingly affluent populations[160]. This framework shows many ways in which nutrients, water, and organic matter can contribute toward augmenting food production.

(a) Paths to enhance supporting services

Input (resource or ES)	Enhanced supporting services: Nutrient cycling	Water cycling	Soil formation	Primary production
Nutrients	Land application of N, P, K			
Water		Effluent discharge		
Organic matter	Land application of carbon		Land application	
Nutrient cycling				Plant nutrient uptake
Water cycling				Plant water uptake
Soil formation				Favorable soil
Erosion control	Reduced soil nutrient losses	Improved infiltration	Reduced soil losses	Stable soil conditions

Matrix structure

	Enhanced ES
Input resource or ES	Mechanism(s) through which the ES is enhanced by the input resource or ES. Note that enhanced ES can act as intermediates, becoming inputs to subsequent steps in pathways that link resources and ***final benefits***)

(b) Paths to enhance regulating services

Input (resource or ES)	Enhanced regulating services: Erosion control	Water regulation	Water purification	Pollination	***Disease/pest regulation***	***Air quality***	***Climate regulation***
Water		Aquifer, stream recharge	Contaminant dilution				
Organic matter	Improved soil structure						Phototrophic (algal) treatment
Nutrient cycling			Reduced aquatic discharge				Soil carbon sequestration
Water cycling		Flood control, wetland storage	Contaminant dilution	Pollinator habitats	Reduced standing water and vector breeding		
Soil formation	Replenish eroded soil	Improved water storage capacity	Soil filtration	Pollinator habitats			Soil carbon sequestration
Primary production	Improved soil and root structures	Improved water retention	Contaminant uptake	Pollinator habitats		Contaminant uptake	Carbon storage
Erosion control		Runoff control, improved infiltration	Reduced solids contamination	Pollinator habitats			Soil carbon sequestration
Water regulation	Reduced runoff			Pollinator habitats	Reduced standing water and vector breeding		
Water purification				Pollinator habitats	Pathogen reduction		
Fuel	Reduced biofuel crop harvest					Reduced fossil fuel use	Reduced fossil fuel use

(c) Paths to enhance provisioning services and biodiversity

Input (resource or ES)	Enhanced provisioning services: ***Fresh water***	***Fuel***	***Food***	***Fiber***	***Ornamental resources***	***Biochemicals***	***Enhanced biodiversity****
Water	Direct reuse	Thermal energy recovery					
Organic matter		Biogas, combustion					
Soil formation		Soil medium	Soil medium	Soil medium	Soil medium	Soil medium	Soil habitat
Primary production		Increased biofuel crop production	Increased food and feed crop production	Increased fiber crop production	Increased ornamental crop production	Plant production	Plant production
Water regulation	Indirect reuse						Aquatic habitats
Water purification	Indirect reuse						Aquatic habitats
Pollination		Crop reproduction	Crop reproduction	Crop reproduction	Crop reproduction	Plant reproduction	Plant reproduction
Climate regulation	Reduced potential for water stress	Alternative biofuels (e.g., from algae)	Reduced climate change effects	Reduced climate change effects	Reduced climate change effects	Alternative bioproducts	Reduced species extinction
Fresh water		Irrigation	Irrigation	Irrigation	Irrigation	Irrigation	Aquatic habitats
Fuel			Reduced biofuel crop area	Reduced biofuel crop area	Reduced biofuel crop area	Reduced biofuel crop area	Reduced monocropping

Figure 5.3. Conceptual maps of potential links between resources from sanitation and ecosystem services. Each matrix presents links between recoverable resources (or intermediate ecosystem services previously linked with resources; resources are shown in black boxes) and (a) supporting services (red boxes), (b) regulating services (blue boxes), and (c) provisioning services (green boxes). Each link within the body of the matrix indicates one or more examples of specific mechanisms through which the input (left-most column) may enhance the ecosystem service in question (top row). This general structure, and the possibility for the enhanced service to act as an intermediate (becoming an input to a subsequent step in a service pathway), is described in the upper-right key. In many cases, resources may connect with final services that are directly valuable to humans (e.g., food) through pathways that include one or more intermediate services (e.g., nutrient cycling, erosion control). Services in bold/italics represent those likely to be of direct value to human populations ("final services"). *While biodiversity is not an ecosystem service, we include it separately to emphasize its critical role in ecological functioning, underpinning many services and Sustainable Development Goals.

Finally, we include biodiversity in the framework as a separate category (distinct from ES) to emphasize its role underpinning many (if not all) ecosystem services[245] and supporting progress toward several SDGs, including those related to poverty, hunger, health, and water[88,225]. Recoverable resources may indirectly enhance biodiversity through various pathways, including by creating or regulating habitats and reducing pressures for land conversion to agriculture (e.g., through diminished need for biofuel crops)[64].

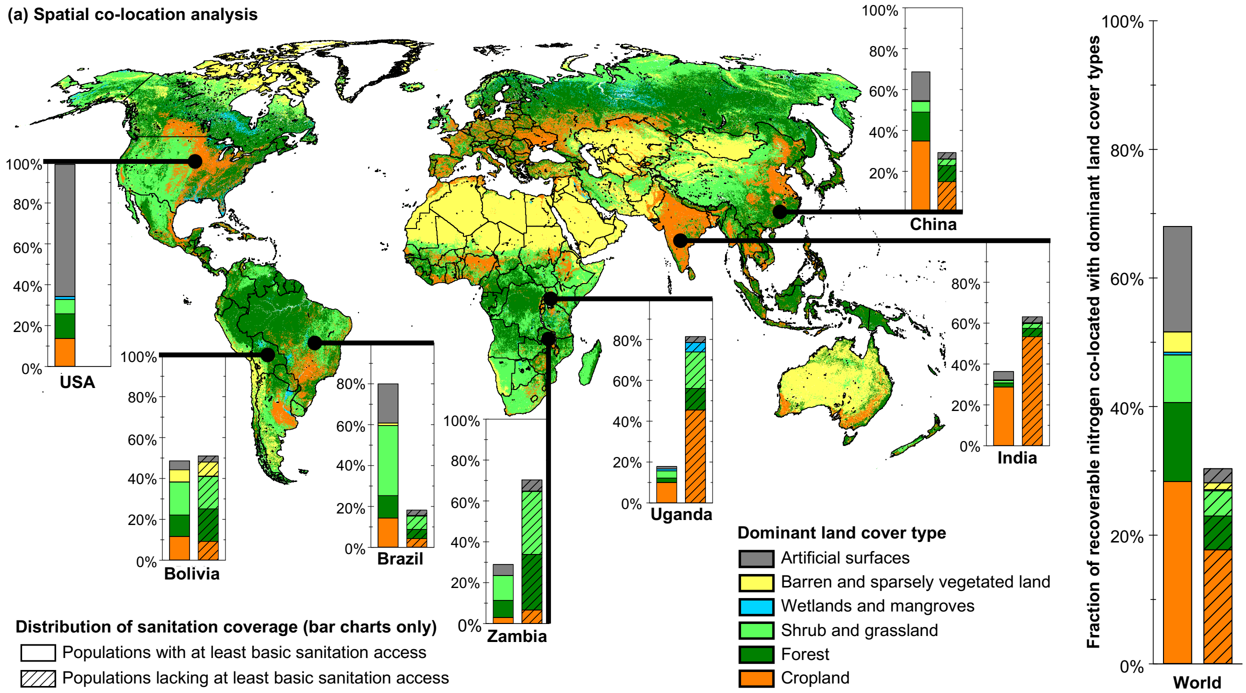
Factors affecting practical feasibility and utility of potential linkages

Numerous issues will affect the feasibility of leveraging these potential links and pathways between resource recovery from sanitation and ecosystem services. Many of the relevant issues, relating to factors such as local topography, soil conditions, climate, health, fertilizer and energy markets, water resources, and socio-cultural norms, will be highly dependent on local circumstances. Here, we begin to examine three potentially critical issues: (i) spatial co-location of recoverable resources and ecosystems; (ii) sanitation and recovery technology choice; and (iii) financing mechanisms.

Spatial co-location. The viability of establishing linkages will likely depend upon the spatial alignment of recoverable resources with the ecosystems those resources could enhance[202]. Being able to recover nutrients, water, or organic matter, and knowing how the recovered products might be used to enhance a given ecosystem, might not be particularly useful if that ecosystem does not exist nearby. Organic matter, for instance, tends to be relatively bulky, and transporting it long distances may not be logistically, energetically, or economically feasible[202]. Therefore, we performed a preliminary analysis to characterize co-location of resources (using nitrogen as a representative example) and dominant land cover (LC) types[246] in 2010 (Supplementary Methods; Supplementary Figures 1-2; Supplementary Tables 2-3), acknowledging this issue will also depend on several local characteristics we cannot capture. We note this co-location analysis is most relevant for nutrient and organic matter recovery (excretion of these resources is related to protein and caloric intake, and they will follow similar spatial trends)[167,202]. Additional global datasets of wastewater generation, treatment, and use (which are less complete[247]) would be needed for a water-focused co-location analysis, providing potential avenues for future study.

Globally, our estimates suggest nearly half of recoverable nitrogen is co-located with cropland-dominant areas (i.e., located in grid cells where cropland represents the largest fraction of total area; Figure 5.4). Indeed, this spatial analysis, our conceptual framework, and existing literature reveal agriculture to be potentially fruitful ground for developing linkages between resource recovery and services such as food provisioning or erosion control, especially in countries where agriculture is a major contributor to gross domestic product, employment, and/or development initiatives (e.g., Uganda, India)[248]. Among populations who already have basic sanitation access, agricultural reuse may necessitate upgrading existing sanitation systems or incorporating additional recovery processes (e.g., to generate concentrated nutrient products), while resource recovery could be an integral factor in the design of new systems for populations without basic sanitation. Areas dominated by forests and shrub/grasslands also align with substantial fractions of resources recoverable from populations both with and without basic sanitation, offering opportunities to enhance services that regulate climate, water, soil, or air quality, particularly in countries such as Bolivia, Brazil, or Zambia.

Approximately 20% of resources are co-located with artificial surfaces (e.g., urban areas, where sanitation coverage tends to be higher[224]). Globally, most people now reside in urban areas (especially in high-income countries such as the United States)[116], although not all urban residents live in grid cells dominated by artificial surfaces. In urban centers, centralized resource recovery might employ technologies generating resource-dense nutrient products to increase the feasibility of transport to natural areas[202], while local water reuse could enhance fresh water provisioning or ornamental crops grown in urban green spaces. Wastewater facilities may also recover chemical or thermal energy to offset energy requirements of treatment[17].



(b) Connections across land cover types, ecosystem services, and resources

Land cover type	Examples of relevant ecosystem services	Examples of resources to recover: Nutrients	Organic matter	Water
Artificial surfaces	Air quality, climate regulation, fresh water, fuel, nutrient cycling, water cycling, water purification, water regulation	Concentrated products	Biogas	Direct/indirect reuse, thermal energy
Barren and sparsely vegetated land	Climate regulation, erosion control, fresh water, soil formation, water cycling, water regulation	Sludge	Sludge	Direct/indirect reuse, aquifer recharge
Wetlands and mangroves	Fresh water, fuel, nutrient cycling, pollination, water cycling, water purification, water regulation	Aqueous discharge	Biogas	Flow regulation
Shrub and grassland	Climate regulation, erosion control, fiber, pollination, primary production, water purification, water regulation	Sludge	Sludge	Stream recharge
Forest	Air quality, climate regulation, erosion control, fuel, primary production, water purification, water regulation	Sludge	Sludge, biogas	Stream recharge, thermal energy
Cropland	Disease/pest regulation, erosion control, food, fresh water, nutrient cycling, pollination, soil formation, water cycling	Conc. products, sludge, irrigation	Sludge	Irrigation water, thermal energy

Figure 5.4. Co-location of recoverable resources from sanitation and land cover types. A global dataset of dominant land cover (LC) types[246] (i.e., the LC accounting for the largest portion of each 0.5 arc-minute grid cell, as shown in the map) was overlaid with global distributions of recoverable nitrogen from populations with and without basic sanitation in 2010 (Supplementary Methods, Supplementary Figures 1-2)[167], to estimate the quantities of nitrogen co-located with each dominant LC (i.e., in the same grid cell). Bar charts show results from the world and seven countries (a), illustrating a variety of co-location distributions (see Supplementary Table 4 for quantitative results from an uncertainty analysis including expected, low, and high nitrogen excretion and recovery scenarios in all countries). This analysis is based on the premise that resource recovery may be simplified by focusing on enhancing ecosystem services in a single, locally dominant LC type, as suggested by the examples presented in alphabetical order in (b). However, places with greater heterogeneity in LC may benefit local actors by offering flexibility, a quality not captured in our analysis. National administrative boundaries that provide the base of the map in (a) were taken from the Gridded Population of the World (Versions 3 and 4)[216,217].

The considerable degree of variability across countries (Figure 5.4; Supplementary Table 4) may impact national policies if governments wish to enhance ES through resource recovery. Certain countries or regions may find the most benefit (or may streamline the reuse process) by aiming to connect recoverable resources with specific ecosystems or land cover types (e.g., concentrated nutrient products applied to cropland to enhance nutrient cycling and food provisioning, or sludge application on forest soils to control erosion and regulate climate through soil carbon storage; Figure 5.4). However, unlike previous work focused specifically on matching agricultural nutrient requirements[202], the co-location analysis presented here did not consider whether some recoverable resources would oversaturate local ecosystems and be better utilized in other, more distant locations. We also note that, when integrating multiple datasets representing different data categories, sources, procedures, and levels of variability, results may be associated with a considerable degree of uncertainty (e.g., see Supplementary Table 4 for results of three scenarios encompassing expected, low, and high nitrogen excretion and recovery potential). Generally, we can conclude from our findings that co-location characteristics of recoverable resources and ecosystems may vary widely across different contexts and will likely affect decision-making.

Sanitation and recovery technology choice. After identifying local ecosystems that could benefit from resource recovery, developing sanitation systems capable of recovering resources to enhance relevant services requires knowledge of the products generated by different technologies. A given product will contain a particular combination of resources and properties, and focusing on its recovery will enhance certain ES but may preclude other benefits. In forest ecosystems, for example, there may be a tradeoff between land application or energy generation from organic matter. Organics can be anaerobically digested to produce biogas, potentially offsetting some need for firewood or other biofuel crops, thereby conserving forests. However, anaerobic digestion consumes much of the organic matter in sludge, potentially reducing the residual's beneficial impacts if land applied. Sludge application can enhance a host of services (e.g., erosion control, climate regulation through soil carbon sequestration, primary production) in forest ecosystems while simultaneously reducing the need for engineered processes to reduce organic solids. Contextual factors such as local energy needs and markets, distances from sanitation facilities to ecosystems of interest, and available technical capacity will likely influence decisions and preferences regarding which approach to take (or how to efficiently combine multiple service pathways).

As another example, nutrients might be recovered in sludge, treated wastewater, or concentrated products (e.g., struvite, ammonium sulfate), all of which can enhance services such as food provisioning, nutrient cycling, and water purification. Each option may be more appropriate under a specific set of local conditions. In urban areas with limited local opportunities for soil application, concentrated product recovery may be most feasible (Figure 5.4), as it reduces the burden of transport over long distances[202]. Conversely, if cropland or aquatic ecosystems are nearby, recovering nutrients in treated water could provide opportunities for agricultural irrigation or wetland discharge, cycling nutrients to terrestrial or aquatic plants. Again, process and product selection will likely depend on factors including the local distribution of ecosystems, soil conditions, and resource availability. Using the conceptual framework to identify and understand contextually relevant products, services, and pathways will be crucial in determining how resource recovery can contribute to conservation efforts, when certain products or strategies might preclude other options, and the relative value of each possibility.

Financing mechanisms. Crucial to the success of any effort linking resource recovery with ES will be the development of efficient financing and management structures that enable households or communities to implement resource recovery technologies and use recovered products. Resource recovery systems may be associated with larger initial investments than basic conventional sanitation (e.g., pit latrines)[158], but longer-term benefits could make them more economically viable over time, especially if additional ecological services can be valued along with more typical agricultural, water, and energy products. Existing commodity markets provide a useful starting point for pricing of products such as agricultural nutrients and energy. For example, a study in 2011 reported a struvite market price of approximately $\$0.53 \cdot kg^{-1}$ in Nepal, comparing favorably with production costs ($\$0.23\text{-}0.56 \cdot kg^{-1}$) if an inexpensive, locally-available magnesium source is used[157]. In other cases, strategies such as payments for ecosystem services (PES) may be appropriate.

Where traditional market mechanisms do not encourage environmental protection, PES (which have increased considerably in recent decades[227,249]) can act alone or in concert with other mechanisms (e.g., carbon pricing) to incentivize interactions that maintain and derive benefits from ecosystems. Such mechanisms provide actors considerable freedom in designing programs that cost-effectively manage local ecosystems. Resource recovery can provide products of direct value (e.g., struvite), creating a potential market for user-financed payments (i.e., where direct beneficiaries of ecosystem services pay)[227]. Alternatively, governments might finance resource recovery in connection with policies supporting universal sanitation access, particularly if they see additional benefits related to agriculture, energy, water supply, or environmental protection. For example, appropriately valuing natural capital to reduce negative externalities (e.g., water pollution)[250] may further incentivize strategies such as nutrient recovery. This approach could integrate multiple ES pathways and financing mechanisms, supplementing the agricultural value of increased crop productivity from application of nutrient products[78] with the benefits of improved water quality from reduced nutrient discharges (Figure 5.2).

The high contextual variability of PES and current lack of payment schemes related to resource recovery inhibit a comprehensive global perspective on the costs and benefits of these potential mechanisms. Conservation programs to prevent deforestation, for example, are reported to pay $\$1.25 \cdot ha^{-1}$ in Guyana and $\$35 \cdot ha^{-1}$ in Costa Rica[251,252]. Similar strategies may offer subsidies to supplement the market price of biogas from anaerobic digestion or other energy recovery products in an organic matter-fuel provisioning service pathway (Figure 5.2) that offsets firewood use from forests. In Indonesia, an erosion control program for coffee farmers paid $\$172 \cdot ha^{-1}$ on average to employ measures such as infiltration pits and vegetation strips[253]. Future programs along these lines might incorporate land application of sludge (perhaps also combined with offsets from carbon sequestration[234]; Figure 5.2), although valuation of sludge would need to consider local characteristics and how much land area a given quantity might affect. These examples suggest the need for future research into the potential of new sources of financing to capitalize on the potential value of underexplored linkages between resource recovery and ES[250].

Furthermore, the unique nature of resource recovery may alleviate concerns sometimes associated with PES. For example, payments often benefit land owners while excluding those without secure land tenure[251], but recoverable resources are produced by everyone. Nevertheless, equity issues surrounding safe access to these resources, as well as the technologies and institutions available to different groups, still remain. Further, questions surrounding resource ownership may also require resolution (e.g., when landlords own sanitation systems used by renters). Generally, though, recovery may provide a distributed source of resources, potentially leading to innovative arrangements where some generate resources and sell them to others who use them to enhance ecosystem services and are compensated by conservation-minded actors. These relationships might offset or prevent employment losses sometimes resulting from PES programs (e.g., those aimed at reducing agricultural or logging activities)[251]. By offsetting such losses, novel sanitation-related PES schemes might provide income opportunities for people to participate in circular economies around sanitation systems.

Pathways forward

This work develops a conceptual framework describing pathways through which resource recovery from sanitation can link with and enhance ecosystem services, while also characterizing certain critical factors (spatial co-location, technology selection, financing) that may affect the practical realization of these connections. Broadly, our findings suggest substantial but underexplored potential to integrate the sanitation and ES fields, specifically regarding resource recovery's possible contributions to surrounding ecosystems through and beyond agricultural nutrients, water, and energy. Integrated system design that couples resource recovery and ecosystem services may represent a synergistic strategy to increase the value of sanitation systems and balance societal needs with ecosystem functioning. Exploring such linkages is especially timely as efforts to achieve the UN Sustainable Development Goals are well underway and the international community seeks to develop a post-2020 framework for biodiversity conservation[254].

However, the impact of many potential connections between sanitation and ES remains uncertain. Quantitative global estimates of potential nutrient and energy recovery, and prospective improvements in access to agricultural inputs and household energy, exist in the literature[29,167], but many other impacts remain unexamined. To properly integrate these linkages into design and decision-making tools, future research should quantitatively estimate various types of ecological impacts and their potential value to ecosystems and human populations across different contexts. Such efforts will likely require greater collaboration among policy-makers, practitioners, and researchers from both fields, along with appropriately-timed engagement with local stakeholders. Ultimately, they would facilitate the development of integrated models for the sustainable design of sanitation systems and conservation strategies, including life cycle environmental impacts and financing structures.

Overall, this work advances knowledge of how research and policy efforts to link sanitation and ES could enhance sustainable development goals relating to environmental protection, human health, and economic well-being. However, it is important to temper expectations regarding the potential for resource recovery to dramatically amplify conservation efforts. Available resource quantities might be relatively small compared with the needs of extensive ecosystems. Additionally, maintaining intact landscapes can provide greater value than restoring degraded landscapes[255]. As recoverable resources may often be used to renew ecosystems after services have been degraded (e.g., organic matter application to reduce erosion and restore depleted soils), strategies focused on conserving intact ecosystems may generate greater conservation benefits than efforts to recover resources.

Finally, as with any multidimensional issue, considering tradeoffs in decision-making is critical. While resource recovery can provide households or communities with a considerable degree of freedom regarding how resources are recovered and used, a scenario where each household follows its own path may not generate impactful benefits on a larger system-wide level. Some regional uniformity may be necessary for developing efficient systems. Cooperation and consensus-building may be required to develop collective sanitation strategies[44], aggregating the individual impacts of many households to produce broader-scale change. Developing appropriate institutions to govern these strategies and relationships, and understanding how these systems may influence power dynamics between stakeholder groups[97], will be central in ensuring that resource recovery does not create unforeseen social challenges or jeopardize human well-being.

CHAPTER 6: A SOCIAL-ECOLOGICAL SYSTEMS FRAMEWORK CONCEPTUALIZING SANITATION AS A HUMAN-DERIVED RESOURCE SYSTEM[e]

Introduction

Globally, over two billion people lack basic sanitation access, and even more may be using systems that do not safely and sustainably manage human excreta[69,71]. Simultaneously, planetary resource use is unsustainable on multiple fronts (e.g., discharges of phosphorus and anthropogenically-fixed reactive nitrogen)[3,64]. Recovery of resources (e.g., nutrients, organic matter, water) from sanitation has the potential to offset some use of resources such as commercial fertilizers and/or improve access for populations in resource-limited settings[167]. However, progress toward the Sustainable Development Goal target of universal sanitation coverage by 2030 remains limited in many places, with numerous countries having coverage below 95%. Of these, only 11% are on track to achieve universal coverage[69], and failure rates of sanitation systems in resource-limited communities are high[37,256].

A range of social, economic, environmental, and political challenges, many of which extend into realms beyond technological performance, can hinder sanitation and resource recovery efforts[19,44,156,257]. Recognition of this fact has led sanitation research to become more interdisciplinary and incorporate greater stakeholder involvement. Consequently, many studies have presented and applied various models, tools, and approaches to explicitly link sanitation with related systems (e.g., agriculture, energy, water) and examine multiple dimensions of sustainability to support participatory decision-making[19,41,44,46,49,84,86,90,96,138,257–264].

This accumulation of research has contributed to a more inclusive and holistic vision for sanitation in various contexts. However, without a common set of core variables to guide the development and application of models, tools, and theories – essentially, without a common,

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[e] This chapter is in preparation to be submitted for publication. Accordingly, all Supporting Materials referenced in this chapter are included in full in Appendices E (general), F (household survey results), and G (research approvals).

interdisciplinary vocabulary elucidating the many potential factors and interactions related to sanitation – scholars may focus on their own interests without considering other characteristics that may be equally or more important in determining outcomes[94,265]. No single model or approach can account for all potentially relevant information, and particularly when studies use differing terminology or do not acknowledge factors that other disciplines may consider critical, placing various disciplines and approaches in conversation with one another can be challenging. As a result, new knowledge may remain isolated[95]. The plethora of tools that have been created might even engender confusion among practitioners uncertain of which one(s) to apply in their situation. Therefore, researchers, decision-makers, and local stakeholders need systematic ways to frame thinking about sanitation and resource recovery across contexts, scales, and disciplines – holistic frameworks built on experience from diverse contexts that establish core structures of variables and relationships for developing more specific models and theories, for understanding how sanitation fits into its broader context, and for aiding stakeholder engagement.

Efforts to systematically analyze coupled human and natural systems (CHANS) have been underway for well over a decade[266,267]. Scholars have now developed a systematic, generalized social-ecological systems (SES) framework[94,95,268,269] and tested it in many contexts[265,270–276]. It emerged from studying the governance of common-pool resources (e.g., forests, irrigation systems, fish stocks)[277]. Generally, common-pool resources are characterized by two key traits: high subtractability (use of the resource diminishes the remaining supply) and low excludability (it is difficult to bar actors from consuming the resource)[272]. As such, governance and management of these resources often require collective action among local communities, and the SES framework offers a nested, multi-tier structure in which to conceptualize, classify, and study these systems from multiple perspectives in diverse contexts[94,95,269,271,272].

When sanitation systems are viewed as potential sources of recoverable resources, they appear similar to common-pool resource systems, while having certain distinctive characteristics. Specifically, they are generated by all people but are only safely available to those with access to appropriate technologies and/or markets (imparting a degree of excludability), and sanitation management, treatment, and recovery strategies may alter the characteristics of these resources. The multi-dimensional and multi-scalar relationships between sanitation processes and broader contextual conditions (e.g., government policies, community-scale priorities, household-level practices)[41,44,46,96,263,278] suggest the need for overarching, modular frameworks to support greater understanding and cooperation among diverse stakeholders[39]. The SES framework may be adapted to conceptualize sanitation as a human-derived resource system, providing a structured, generalized understanding of sanitation's existing and potential functions within broader communities.

Accordingly, the objectives of this study are (i) to develop a conceptual framework for sanitation and resource recovery systems based on insights from SES research, and (ii) to illustrate the utility of the framework through application in a specific resource-limited context. Based on literature surrounding the SES framework, sustainability analyses, and sanitation decision-making, we modify the generalized SES framework to consider the unique characteristics of sanitation and resource recovery systems. To illustrate how this process can enable researchers, decision-makers, and stakeholders to collaboratively elucidate sanitation possibilities in particular settings, we integrate quantitative modeling with household surveys and stakeholder discussions to apply the framework in Bwaise, a densely-populated informal settlement in Kampala, Uganda.

Overall, this work represents an important step toward more holistic, interdisciplinary understanding of sanitation and its functions in different local settings around the world. Critically, the framework remains open for future development. One group or study cannot adequately account for all possibilities, and future work is needed to iteratively develop and apply the framework, improving it over time. As it grows, it will become a better tool for understanding sanitation, systematically studying what variables help ensure successful and sustainable systems, and informing participatory scenario development and decision-making.

A social-ecological systems (SES) framework for sanitation and resource recovery

Following the approach of the SES community, we define frameworks as being categorically distinct from models or theories[269]. Models and theories describe and examine specific aspects within a broader topic, often requiring simplifying assumptions to control potential variations in other parts of the system. In contrast, a framework provides a common vocabulary of variables and relationships as a foundation for the development of theories and models within an overarching structure[269,277]. Essentially, models and theories can integrate into specific sections of a framework to study a subset of relevant variables and relationships, while the framework ensures awareness of other system features. By enumerating general classes of key variables applicable to systems across diverse contexts, frameworks help to avoid two extremes: excessive generality (offering little meaningful content) and excessive precision (offering little applicability or accuracy when applied in different circumstances)[269]. Simultaneously and in reciprocal dialogue, ongoing model and theory development suggests ways in which frameworks can be improved to provide greater clarity and additional information regarding key variable classes[269].

Adapting the overarching structure of the SES framework. The general SES framework incorporates a nested structure to conceptualize relationships between variables, providing mechanisms to understand a complex whole and facilitate research at various levels of specificity[94,95,269]. It includes five first-tier variables representing the core subsystems that interact within the overall SES: resource systems, resource units, actors, governance systems, and focal action situations (where interactions occur, decisions are made, and outcomes generate feedbacks to other variables)[95,269,271]. Additionally, two broader first-tier variables (related social, political, and economic settings; related ecosystems) represent the general context in which the SES operates[95]. Each first-tier variable contains multiple levels of additional variables. Most studies concerning the general framework have established several second-tier variables within each first-tier component[94,95,269]. A few studies have proposed general third-tier variables nested within second-tier components[269], further expanded lower tiers in a specific context[270], or built out particular lower-level components for illustrative purposes[273]. The framework has undergone continued development and extension since its inception for more general application to contexts beyond common-pool resource systems, including those with distinct technological components (e.g., energy infrastructure)[271,272]. Additionally, related extensions such as the coupled infrastructure systems framework provide alternatives to SES nomenclature and analysis units, for example by redefining system components as different forms of infrastructure (e.g., built, natural, institutional, human)[279].

Inspired by the authors' experience working on sanitation issues in a resource-limited setting (an informal settlement in Kampala, Uganda) and based on a review of existing literature from multiple fields (SES, sanitation decision-making, general sustainability analysis), we modified the first-tier configuration of the existing framework in three major ways to create the overarching structure for the sanitation SES framework (Figure 6.1). First, we define human-derived resource systems as including resource units (e.g., nutrients, organic matter, water) and actors (who generate resource units through their use of sanitation facilities). Second, reuse systems have been added to describe how recovered resource units may be employed. Finally, decisions and outcomes around the selection and management of sanitation technologies now form the keystone of the framework. As with the more broadly-defined focal action situations of the generalized SES framework, this subsystem represents a point of interaction across the other core subsystems. The attributes of actors, governance structures, resource units, and reuse options, as well as broader social, political, economic, and ecological characteristics, will affect the viability of sanitation opportunities. Alternative perspectives (e.g., the coupled infrastructure systems framework) may also be useful in understanding relationships and interdependencies across subsystems[265,279]. However, human-derived resource systems may cross multiple infrastructure categories and be difficult to define within the coupled infrastructure systems framework, illuminating a key contribution of our sanitation SES framework.

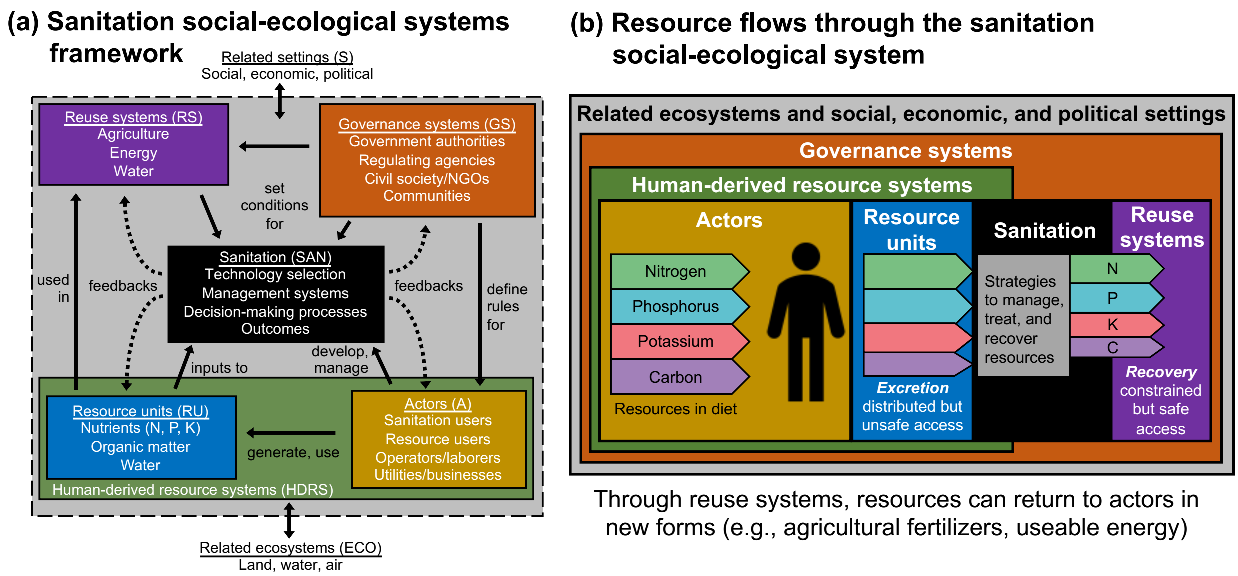


Figure 6.1. Adapting the social-ecological systems (SES) framework to sanitation. (a) The diagram shows the core subsystems (first-tier variables; e.g., sanitation, actors, related ecosystems) of the SES framework adapted to sanitation and includes examples of distinct categories (second-tier variables; e.g., technology selection, sanitation users, water) within each core subsystem. The framework structure can extend beyond the first and second tiers to provide additional layers of detail as needed. A listing of the second-, third-, and fourth-tier variables we have included in the framework can be found in Tables E.1-E.4. (b) Resource flows through the core subsystems of the sanitation SES suggest how the safety and accessibility of resources may change as they move through various stages. Appropriate management, treatment, and recovery strategies can increase safety and minimize risks associated with recovered resources, but these processes may introduce constraints on access related to technology availability, economic resources, and knowledge of sanitation and hygiene. Moving forward, the overall framework can be expanded to include more variables and levels providing additional breadth and depth, with the goal of establishing a comprehensive vocabulary of concepts and relationships to inform decision-making and the development of appropriate models and theories.

Building out multiple tiers of each core subsystem. After restructuring the core subsystems of the SES framework with sanitation and resource recovery in mind, we then expanded each first-tier component to include several variables in multiple lower tiers (Tables E.1-E.4). Generally, we define second-tier variables as distinct categories of core subsystems or common issues crossing multiple categories, while third- and fourth-tier variables represent attributes that are nested within higher levels to provide further detail (while the general SES framework must be flexible and open enough to consider any type of resource system, constraining our version to sanitation alone allows for more specificity at lower levels). However, it should remain general enough to include any sanitation system, encompassing a range of scales, technologies, management strategies, and levels of complexity. Even the fourth-tier variables we include are not detailed enough to fully characterize individual systems. Future studies making use of this framework (or advancing it for specific disciplines, contexts, or technologies) should go further still, defining lower-tier variables appropriate for their applications.

The central sanitation subsystem contains the largest number of lower-tier variables, encompassing the design and management of sanitation facilities, the processes used to reach decisions, and the multi-dimensional outcomes of these decisions (Table E.1). In this framework, we conceptualize a full sanitation system as an integrated value chain of distinct processes (user interface, onsite storage/treatment, conveyance, centralized treatment/recovery, reuse/disposal), each of which must be considered on its own and in relation to its complementary processes when designing contextually appropriate facilities and management strategies[84,264]. Design and decision-making processes themselves may also contribute to the continued functioning of a system. Participatory processes that engage local stakeholders and municipal authorities can increase the likelihood of success by identifying community priorities, developing alternative scenarios, building consensus, and navigating tradeoffs involving different stakeholder groups or value categories[19,37,44,97,256,261,263]. Finally, the outcomes associated with a proposed or implemented system may cross numerous dimensions of sustainability, including economics (e.g., life cycle costs, user fees, resource value, subsidies), environmental impacts (e.g., pollution, climate change potential, conservation), resource efficiency (e.g., materials, energy, water, land sparing), human health (e.g., disease prevalence, risk, nutrition), social acceptability (e.g., regulatory compliance, ownership, user requirements, employment opportunities, equity), and technological robustness (e.g., performance uncertainty, shock sensitivity)[4,41,100,101,111,231,280–286].

All other subsystems feed into these sanitation decisions and outcomes, and implemented sanitation systems create feedbacks that influence the characteristics and possibilities associated with these other aspects. Subsystems concerning resource units and reuse options focus on three general categories of resources recoverable from sanitation (nutrients, organic matter, water) and their potential role in agriculture, energy, and water systems (Table E.2). For each resource category, the framework characterizes generation rates (dependent upon local diets, sanitation system configurations, and recovery technologies), forms of recovery (e.g., nutrient-dense products such as struvite may be easier to transport and store than sludge), product value (e.g., cooking or heating capacity of recovered energy, fertilization and irrigation benefits of nutrients, carbon, or water in agriculture), and existing alternatives (e.g., recovered nutrients may offset fertilizer imports or increase fertilizer access where availability is limited)[15,29,76,157,167,171,202,287]. Likewise, reuse systems define the need for and potential usage of recovered resources (e.g., related to agricultural crop patterns, household cooking needs, existing water use), further characterize various conventional alternatives (e.g., inorganic fertilizer application, fossil fuel combustion, groundwater extraction), and consider the possible impacts of reuse (e.g., economic gains from improved crop yields, greenhouse gas offsets from reduced nitrogen fertilizer production through the Haber-Bosch process)[6,26]. Across all reuse systems, additional factors related to proximity of recovered product supply and demand for reuse, spatial and temporal variations in demand, storage and transport capacity, and actors' ability to acquire and maintain necessary infrastructure for reuse will affect the feasibility of potential recovery strategies[76,202].

Various actors and governance systems may impact sanitation (Table E.3). The involvement of stakeholder groups (e.g., community members using sanitation or recovered resources, system operators and utilities, local government authorities) in planning is a critical element of pathways toward sanitation success[37,256]. The spatial and demographic distributions of users of sanitation systems and users of recovered resources (these two populations may be separate or overlapping), along with these groups' preferences and norms (e.g., cultural beliefs, gender roles), may relate to sanitation system scale and location, market possibilities for resource recovery, and the proximity of supply and demand[146,261]. For sanitation users in particular, dietary intake will influence the quantities of resources being excreted into sanitation systems for potential recovery[29,58,167]. Across all actors, social capital and the properties of social networks may affect levels of trust, willingness to interact with various groups and institutions, motivations for interaction, and the prospects for building consensus[44,288,289]. Additionally, sanitation and resource systems must be compatible with the regulating and enforcement mechanisms of various governing authorities, the property-rights arrangements of communities, and the decision-making, monitoring, and accountability frameworks of implementing organizations[94,95,277].

Finally, these subsystems exist within a broader context defined by social, economic, political, and ecological settings (Table E.4). Social characteristics surrounding demographic trends, cultural norms, and available institutions for human capital (e.g., education and skills training, health care) will affect the distributions, preferences, priorities, and capacities of various actors[47,263,290]. Similarly, political attributes (e.g., forms of government, national and regional stability) may impact workable governance frameworks, collaborative opportunities, jurisdictional boundaries or conflicts, economic volatility and adaptive capacity, barriers to infrastructure development, and the migration, displacement, or marginalization of local populations[38,289,291]. Simultaneously, economic conditions related to international trade networks and foreign investments, access to savings and credit institutions, and availability of resources and raw materials can influence local sanitation and resource markets and incentives, existing infrastructure resilience, and the interest rates and taxes potentially associated with sanitation investments and business opportunities[278,284,292,293].

Sanitation and resource systems also interact closely with local and global ecosystems. Local climate may impact the viability of certain types of treatment systems (e.g., temperature-dependent biological processes, pathogen inactivation via solar radiation)[5,294]. Land use patterns and soil characteristics (e.g., pH, nutrient retention capacity, organic carbon reserves) may affect the recovery products most appropriate in a given context[202,295], while land application of recovered nutrients may enhance ecosystem services[296] and reduce unintended environmental impacts (e.g., eutrophication caused by nutrient discharge into water bodies)[90]. Furthermore, some treatment processes (e.g., conventional activated sludge) can be energy-intensive, and the degradation of bodily waste can emit potent greenhouse gases under certain conditions (e.g., methane in anaerobic environments, nitrous oxide where nitrification/denitrification occurs)[5,281,285].

Together, these subsystems of the sanitation SES framework enumerate a wide-ranging (though not exhaustive) taxonomy of attributes and parameters that may be considered when studying, designing, or modeling sanitation systems and their roles in context-specific scenarios. A study's selection of relevant variables and relationships will depend upon its hypotheses, objectives, and methods. Our goal in developing this framework is to promote consistency and raise awareness of diverse factors that cross disciplinary boundaries – factors that (understandably) may be external to some study designs but that may be equally important in influencing the success and sustainability of sanitation systems. To illustrate the ways in which the sanitation SES framework can inform research and decision-making – both by identifying parameters to include in quantitative modeling approaches and by acknowledging additional (unmodeled) factors – we demonstrate its application in a specific resource-limited setting to begin to study existing and alternative sanitation scenarios. Reciprocally, we note our work in this context helped to inform framework development, and applications in other locations may offer further insight into how to expand the framework moving forward.

Testing the sanitation SES framework

Contextual setting. The growth of urban areas of low- and middle-income countries has placed increasing pressures on sanitation and resource infrastructure[297] (S.S.1.2; this and subsequent alphanumeric designations denote specific framework variables, found in Tables E.1-E.4, relevant to the associated statement). In Kampala, Uganda, onsite sanitation facilities (e.g., pit latrines) serve 90% of the population, with an estimated 37% of these systems being safely managed[298] (A.S.3.1, A.S.3.2). Sanitation patterns tend to be similar in other cities in low- and middle-income countries, particularly in sub-Saharan Africa[299]. Even when safely emptied and transported to a treatment facility, fecal sludge from onsite systems can cause challenges (e.g., high solids, organics, and nutrient loading) that sewage treatment plants designed for more dilute wastewater flows are not prepared to handle[299] (SAN.T.2.3).

In particular, sanitation approaches in informal settlements often fail to meet the needs of already underserved and vulnerable populations, leading to recent efforts specifically focused on fecal sludge management in these contexts[299]. In northern Kampala, the Lubigi Sewage Treatment Plant began operation in 2014 and manages both sewer influent and collected fecal sludge via sedimentation, drying beds, and lagoons (SAN.T.1.4). Dried solids are available for local farmers to purchase as a soil amendment (SAN.T.1.5). Fecal sludge collected from pit latrines is trucked to the plant (SAN.T.1.3) from surrounding communities such as Bwaise, an informal settlement with over 100,000 people[300] spanning three parishes of northern Kampala (S.S.1.1; Figure 6.2). Bwaise is located in a low-lying area formerly classified as a wetland ecosystem[301] (ECO.W.2.4). Most people use shared latrines (serving multiple households), which tend to fill frequently (often in ≤1 year) and may contaminate the shallow groundwater table, particularly when flooding occurs during the wet season[300–302] (SAN.T.1.1, SAN.T.1.2, A.S.3.1). Furthermore, issues of vehicle inaccessibility hinder sludge emptying by vacuum trucks (S.E.5.2), and the contents of some latrines may be discharged into adjacent pits or open drains (A.S.3.2), resulting in additional health concerns[301] (SAN.O.4). Indeed, a settlement profile published in 2014 reported sanitation as a high development priority among community members in two of Bwaise's three parishes[300] (A.S.4, A.S.5).

Data collection. This partial illustrative application of the sanitation SES framework represents a collaboration among Community Integrated Development Initiatives (CIDI), Makerere University (MU), and the University of Illinois at Urbana-Champaign (UIUC) to characterize and assess selected SES components (including resource circularity, economics, and greenhouse gas emissions) related to the existing sanitation system and potential alternative approaches (SAN.D.2) through local data collection, stakeholder engagement (SAN.D.1), and quantitative modeling (SAN.D.3.2; Figure 6.3). As part of a broader effort to characterize water, sanitation, and health conditions within Bwaise, we carried out a household survey across the settlement's three parishes (A.S, A.R, GS.C). Portions of this survey helped us to establish an understanding of local challenges and opportunities related to sanitation and resource recovery.

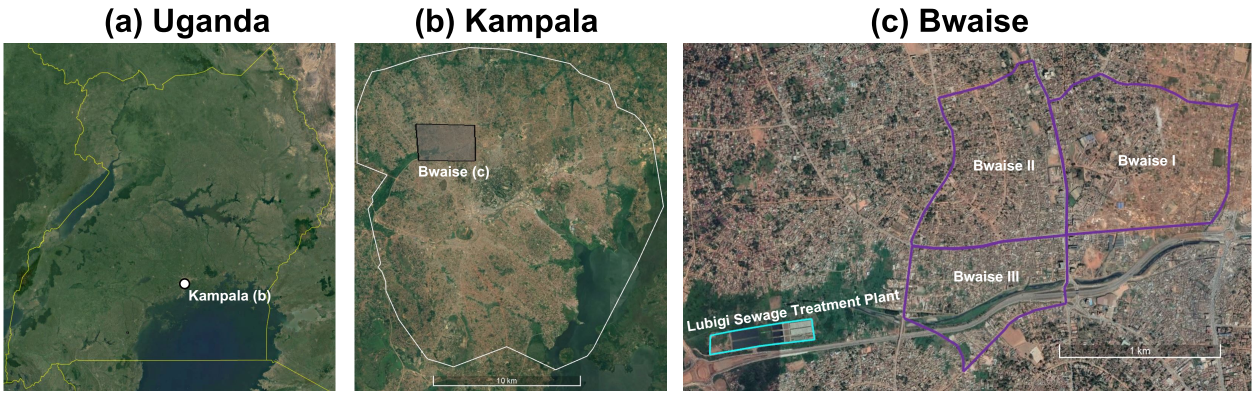


Figure 6.2. Map of the study site within its broader national context, showing Kampala's location in Uganda (a), Bwaise's location in Kampala (b), and the three parishes of Bwaise (c). Bwaise I, II, and III are outlined in purple, and the Lubigi Sewage Treatment Plant is outlined in light blue. Each parish contains ten zones (i.e., villages), and three representative zones were selected from each parish in collaboration with community leaders from all zones to ensure adequate geographic representation across Bwaise.

Each of the three parishes (Bwaise I, II, and III; Figure 6.2) contains approximately ten zones (i.e., villages). CIDI and UIUC representatives engaged with community leaders from all zones to select three representative zones from each parish and implement the survey over a period of nine days in May 2018. The data collection team – consisting of 2-3 CIDI staff, 2-5 UIUC researchers, and ten survey enumerators from MU, all supported by community leaders – visited a different zone each day. Households were randomly selected by enumerators in each zone, and, if the female household head or primary caregiver was present and available to participate, that zone's community leader introduced the enumerator. The community leader was not present while the survey was being conducted, as one leader was supporting and providing introductions for ten enumerators. The goal was for each enumerator to visit approximately ten households per day, resulting in an overall goal of 900 households (with 100 from each zone, representing approximately 4% of the estimated total number of households across Bwaise's three parishes). Across all zones and parishes, enumerators were able to visit 897 households where the female household head or primary caregiver was present, with a total of 844 people consenting and completing the survey (conducted in Luganda, the local language). Participating households were evenly distributed across the nine zones, with each zone contributing 10-12% of the total. The survey, designed in collaboration with social scientists experienced in survey design and interpretation, covered a variety of topics including child health, water, sanitation, energy, agriculture, and diet (full survey instrument included as Survey E.1 in Appendix E). Human subjects research approvals were obtained from UIUC and MU Institutional Review Boards, and the Uganda National Council for Science and Technology approved the study (Appendix G).

The survey focused on female heads of household or primary caregivers because one of its focus areas was designed to characterize child health. As such, nearly all participants were women (92%). Surveyed households contained a median of four people, and the median reported income was 150,000 Ugandan shillings (approximately 40 USD), although over half of respondents reported not knowing their household's monthly income or were not willing to share. Most respondents reported that their household has an electricity connection (82%) and assets such as televisions (64%) and sofas (52%), while fewer reported owning items such as refrigerators (23%) or computers (4%). Approximately half of respondents reported having at least some secondary schooling, but only 10% reported that they had completed secondary school or had begun or completed tertiary education (full results of the household survey are included as a supplemental electronic file, described in Appendix F).

We also interacted with various other key actors to further develop contextual understanding. CIDI's own experience working for several years in this community was invaluable in providing local background and technical expertise. We also held unstructured discussions with stakeholders and decision-makers such as the Kampala Capital City Authority, sludge emptying truck operators, and Lubigi Sewage Treatment Plant employees (A.T, A.U, GS.G, GS.R, GS.N). These discussions focused on topics including existing sanitation practices in informal settlements, alternative management approaches, and water and sanitation funding (photographs showing Bwaise, local sanitation systems, research partners, and stakeholder discussions are included as Photographs E.1 in Appendix E).

Finally, while our modeling of sanitation scenarios (summarized below and in Table E.5; additional details in Sections E.1-E.5) was based in part on survey responses and stakeholder discussions, several additional assumptions were required to develop full simulations. We used relevant literature and work done in similar settings to assign appropriate parameter values and characterize the uncertainty of assumptions (Tables E.6-E.11).

Scenario and model development. Based on collected data, we developed three potentially appropriate sanitation scenarios to model and evaluate. For each alternative, we assessed multi-dimensional outcomes including net life cycle costs (SAN.O.1), greenhouse gas (GHG) emissions (SAN.O.3.4), and the recovery potential (SAN.O.2) of multiple nutrients (nitrogen, phosphorus, potassium; RU.N) and organic content (recoverable as organic matter for land application or energy; RU.O). Cost and emissions estimates incorporated offsets representing potential use of recovered products (detailed modeling procedures are described in Sections E.1-E.5). Health risks are an important factor to consider when studying sanitation outcomes (SAN.O.4), but we do not quantitatively model them in this application, as uncertainties around reductions and risks associated with various pathogens would be too high at this stage to adequately distinguish between alternatives[303–305]. Finally, we note our primary goal in modeling these alternatives was not to definitively select one "best" option. Rather, combining model results with other relevant (but unmodeled) factors can enable local organizations (e.g., CIDI) to present communities and other decision-makers with multi-dimensional information on these alternatives and their tradeoffs (SAN.D.3.2). These actors can then consider contextual priorities to determine the value of each possibility and choose an appropriate path forward (SAN.D.3)[19,44,263,302].

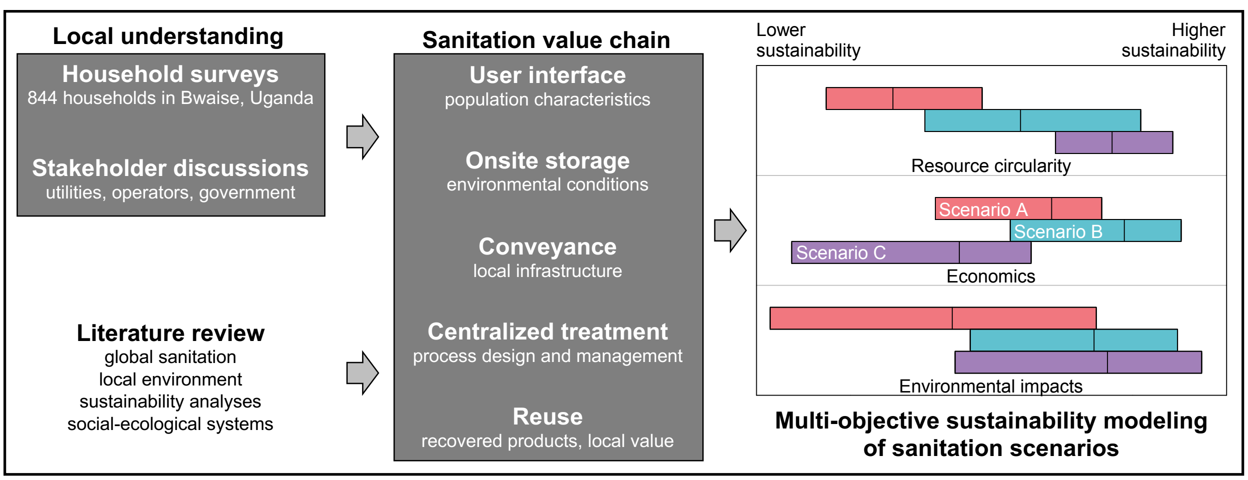
The three modeled scenarios include (Scenario A) the existing sanitation system, (Scenario B) an alternative treatment center proposed by CIDI, and (Scenario C) a container-based sanitation system with centralized treatment similar to the existing system (Table E.5). Consistent with recent literature on the systems nature of sanitation, we employed a modular modeling framework to conceptualize each alternative as an interconnected sequence of processes, from onsite facilities through to centralized treatment, recovery, and potential reuse[84,264] (SAN.T.1). The existing system (Scenario A) includes pit latrines shared by ≥3 households (based on survey results and existing literature[301,302]). When latrines are full, tanker trucks extract the sludge and transport it to a centralized facility. In reality, some latrine sludge may remain unemptied (due to issues such as affordability, truck accessibility, or sludge consistency) or may be discharged into nearby drains[301,302] (A.S.3.2). However, to assess the full potential of the existing system under ideal conditions (and to compare it with the hypothetical alternatives operating under design conditions), we assume all sludge is collected appropriately. Centralized treatment (with a sludge capacity of 500 $m^3 \cdot d^{-1}$) contains sedimentation followed by drying beds for solids management and lagoons (anaerobic and facultative) for liquid management. Following treatment, we assume all dried solids are purchased by local farmers for cropland application. To provide the greatest recovery potential, we also assume liquid effluent is used for crop irrigation, providing additional economic benefit and offsetting emissions from fertilizer production. In reality, plant effluent is discharged into local wetlands, where some crop production does occur (Section E.3).

The scenario containing CIDI's proposed treatment center (Scenario B) employs the same onsite latrines and truck conveyance processes as the existing system. However, a smaller treatment facility now includes a three-chambered anaerobic baffled reactor (overall hydraulic retention time of 1-5 days[306]), followed by unplanted and planted drying beds for solids management and an additional planted bed for secondary treatment of liquids. CIDI estimates an influent sludge flow of 60 $m^3 \cdot d^{-1}$. We assumed solids and liquids are recovered for land application, while digester biogas provides cooking energy (Section E.4).

Finally, in the container-based sanitation scenario (Scenario C), pit latrines would be replaced with container-based facilities, separating and storing urine and feces in small containers that are collected frequently (e.g., twice per week)[96,113,114,305]. This approach requires a reliable collection scheme able to efficiently access facilities in densely-populated settlements with limited road infrastructure (A.S.1.1, S.E.5.2). Manually-operated pushcarts may address these constraints[96,113,114]. While container-based sanitation may include semi-centralized treatment facilities in the settlement (within walking distance for pushcart operators), we conservatively assume containers are transported to a centralized plant farther from collection sites (e.g., the Lubigi plant is 4-5 kilometers from Bwaise). Therefore, this scenario incorporates pushcarts to collect containers and bring them to a nearby truck for conveyance to the plant. We assume a similar treatment approach to the existing system (excluding sedimentation, as liquids and solids are already separated), but we acknowledge that other centralized treatment and recovery processes (e.g., composting, extended storage, struvite precipitation) may be more appropriate for managing desiccated solids and source-separated urine. Use of the existing treatment approach allows us to directly consider the implications of replacing pit latrines with container-based toilets (Section E.5).

Across all scenarios, we employed Monte Carlo analysis with Latin Hypercube Sampling[133] (10,000 simulations) to account for uncertainty around assumptions and model outputs. When conveying numerical results, we report the median value (5^{th}-95^{th} percentiles) from the uncertainty analysis. For each uncertain input parameter (Tables E.6-E.11), we calculated Spearman's rank correlation coefficients[184] to assess the sensitivity of each modeled outcome to that parameter (Section E.1).

Findings. Quantitative modeling results reveal synergies and tradeoffs concerning the costs, emissions, and recovery potentials of the three sanitation alternatives (Figures 6.3-6.4). The existing system (Scenario A) has comparatively low recovery potentials (SAN.O.2) and high GHG emissions (SAN.O.3.4), predominantly caused by pit latrine losses (Figure 6.3). Direct gaseous emissions of methane and N_2O from degradation of excreta during storage in pit latrines contribute 39% (23-53%) of the scenario's total emissions. Generally, direct methane and N_2O emissions (from latrines and centralized treatment) represent 78% (54-90%) of the scenario's total emissions, outweighing other emissions categories (e.g., conveyance, construction and operation of toilets and treatment facilities). These direct emissions are the primary cause behind particularly low recovery potentials of 25% (14-42%) for nitrogen and 8% (4-15%) for chemical oxygen demand (COD, an aggregate measure of organic matter). Leaching from latrines may account for large additional nutrient losses, although leaching rates can vary considerably with local conditions[231,301,307–309] (ECO.L.2, ECO.W.2.4). Regarding system costs (SAN.O.1), the construction of treatment facilities contributes 36% (22-49%) of total costs, while toilet construction represents 30% (18-49%). Short expected lifetimes (e.g., 8-11 years for the treatment plant, which cost over $18 million to build; SAN.T.2.1) make these construction costs particularly important, while low recovery potentials and products with relatively low market value[101] (e.g., <$17·tonne^{-1} of dried sludge; RU.N.2, RU.N.3) lead to revenues that offset negligible fractions of total system costs. However, plant construction was at least partially subsidized by international funding and donor agencies[310] (S.E.3.2), such that served households are not responsible for full system costs (though these subsidies raise questions of sustainability).



Contextual analysis of sanitation as a human-derived resource system informed by a social-ecological systems framework

Figure 6.3. Contextual application of the sanitation social-ecological systems (SES) framework to assess alternative sanitation scenarios in Bwaise, Uganda. Using methods and understanding informed by the sanitation SES framework, we integrated local data collection, stakeholder engagement, and information from the literature to quantitatively model and assess the potential of three scenarios (the existing sanitation system and two alternative possibilities) relative to multi-dimensional outcomes, taking the entire sanitation value chain (i.e., all components in the process sequence from user inputs to treatment, recovery, and potential reuse). Across sustainability dimensions of resource circularity (resource recovery potential), economics (net life cycle costs), and environmental impacts (net greenhouse gas emissions), results revealed potential tradeoffs and synergies associated with transitioning from the existing system (Scenario A) to one of the alternatives (Scenarios B-C). See Figure 6.4 and sections on *Scenario and model development* and *Findings* for further details.

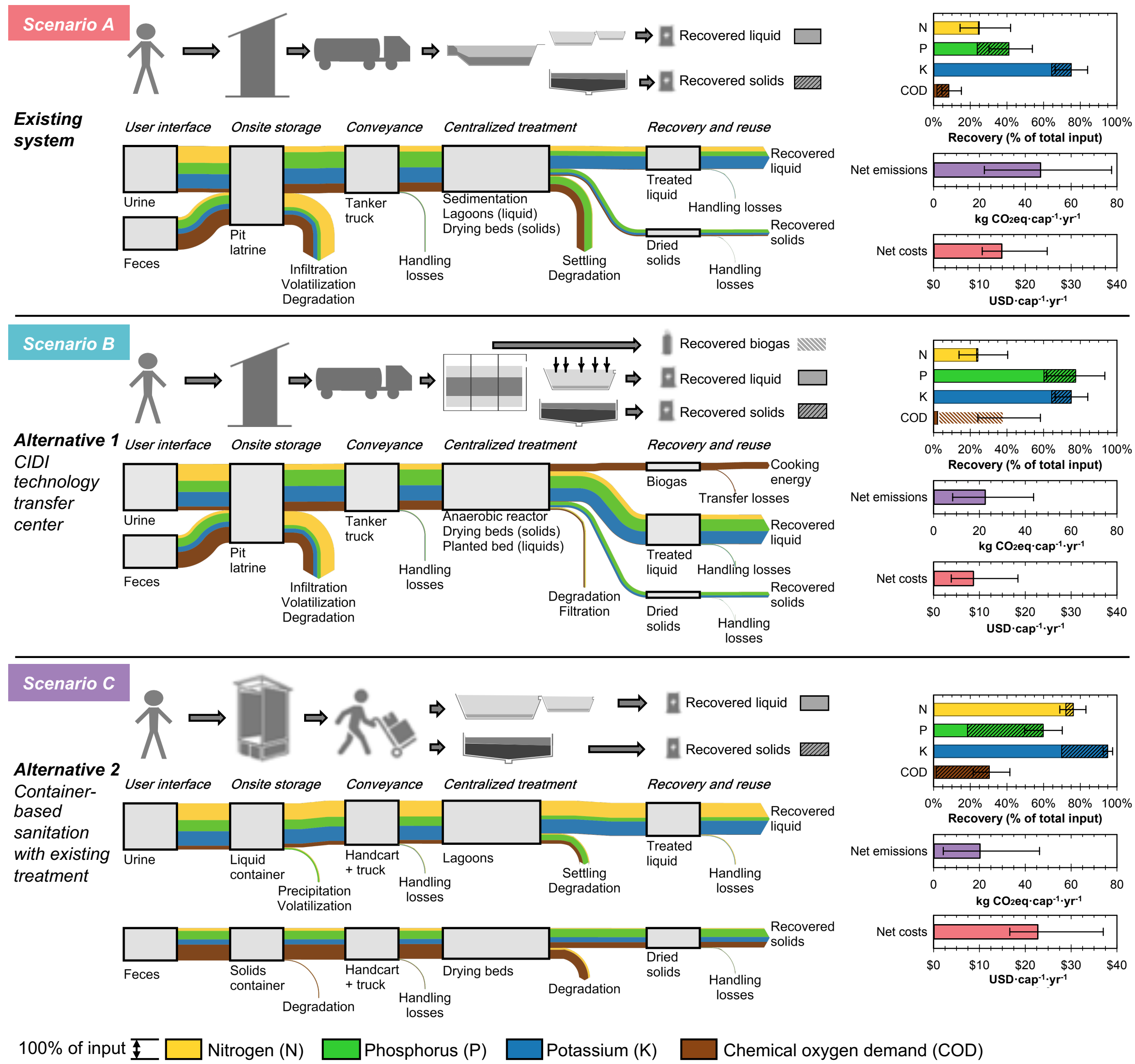


Figure 6.4 Multi-objective comparison of the quantitative outputs (N, P, K, and COD recovery potentials; net GHG emissions; net costs) associated with each scenario. Below the pictorial summary of each scenario, a Sankey diagram shows median flows and losses (relative to initial inputs) of N, P, K, and COD through each stage of the sanitation chain. Bar graphs on the right show overall recovery potentials of each resource (corresponding to the final recovered flows from the Sankey diagram, with recovery in liquid shown with no hatching, recovery in solids shown with black hatching, and recovery in biogas shown with white hatching), as well as total net GHG emissions and costs estimated for each system. Bars reflect median values, while error bars represent 5th and 95th percentile values, extracted from the 10,000 simulations included in the uncertainty analysis.

CIDI's proposed alternative treatment center (Scenario B) appears to perform better than the existing system with respect to all quantitative metrics, exhibiting higher recovery potentials, lower emissions, and lower costs. The treatment center's lower estimated construction cost and longer expected lifetime (50 years) contributes to reducing annual system costs. Substantial resource losses still occur during latrine storage, but different centralized treatment helps to reduce subsequent emissions. Anaerobic processes capture methane emitted from degrading organic matter, mitigating emissions and producing biogas useful as cooking fuel (RU.O.2, RU.O.3), while most nutrients remain in recovered solids and liquids (RU.N.3). Revenues from recovered products are higher than in the existing system, offsetting 26% (11-54%) of total costs, with the sale of biogas (assumed to be similar in value to locally-sold propane gas) representing most of this benefit. Based on survey responses, households currently spend a median of 33% of income to purchase charcoal for cooking (RS.E.2.1, A.S.1.3), and a preliminary analysis comparing the energy content and heating efficiency of biogas to charcoal suggests biogas may be a considerably less expensive alternative (RS.E.1.1, RS.E.3.1). However, a household's startup costs (e.g., biogas stove and tank) may present a barrier, while the logistics of bottling biogas to create a marketable product may create new challenges and represent additional costs for sanitation providers. CIDI and other local organizations are also pursuing the possibility of making fuel briquettes from recovered sludge (rather than selling it as a soil amendment), which may represent another income stream but may entail additional processing costs.

The container-based system (Scenario C) has the highest nitrogen and potassium recovery potentials across all three alternatives and low emissions similar to those in Scenario B, highlighting the important role of decentralized toilet facilities in the full sanitation chain. As urine and feces are only briefly stored in impermeable containers before being transported to centralized treatment facilities, leaching losses are eliminated (reducing potential for groundwater contamination) and emissions are minimized. Therefore, more nitrogen and potassium remain in final products (despite some treatment losses), while COD recovery in sludge is much higher than in the existing system (Scenario A). However, decentralized storage may entail some phosphorus losses, as conditions in undiluted stored urine can promote precipitation of phosphorus-containing minerals including struvite and hydroxyapatite, a portion of which may form a scale on container walls that is difficult to remove[61,311–313].

Despite the potential environmental and resource benefits of this system, however, its costs are estimated to be the highest of the three scenarios. As defined here, the container-based option combines the more expensive existing treatment facility (discussed previously), container-based toilets estimated to have higher capital and operating expenses than pit latrines, conveyance involving both pushcarts and trucks that represents 37% (21-50%) of total costs (indicating the importance of transport in the sanitation service chain[299]), and low-value recovery products. Additionally, discussions with community leaders revealed that the introduction of new toilets may heighten questions of ownership, particularly around landlord-tenant relationships, which will require resolution to equitably distribute benefits.

Generally, this result suggests that the potential of container-based sanitation to be economically viable[96,113] may necessitate wholesale (and potentially disruptive) changes to sanitation systems (i.e., replacing pit latrines while keeping other elements of the sanitation chain constant may not be effective). Semi-centralized treatment located nearer to communities would reduce the need for truck transport. Alternative treatment processes (e.g., co-digestion of solids, struvite precipitation from urine) may provide options that are more suited to source-separated materials and better able to generate higher-value products that could substantially increase revenues (RU.N.2), though advanced recovery possibilities may add substantial capital or operating costs (e.g., magnesium availability for struvite precipitation; S.E.5.1)[157,314,315].

Alkaline products such as struvite may be particularly beneficial for farmers in Uganda, offering beneficial pH adjustment to acidic soils common in the country[295,301] (ECO.L.2.1). Although few survey respondents practice agriculture, nutrient-intensive crops such as maize and matooke are commonly consumed by households in Bwaise, suggesting nearby farmers likely grow these crops and may benefit from better nutrient availability (RS.A.1.1)[167]. Existing (albeit low-value) markets for dried sludge, along with an analysis of the co-location of crop nutrient demands and potential urban nutrient recovery, indicate the local proximity of farms may not constrain reuse of bulkier products such as compost or anaerobically digested sludge[202] (RS.G.4, A.R.2). Especially for organic-rich products, innovative mechanisms that link reuse with ecosystem services (e.g., erosion regulation, carbon sequestration in soil; ECO.ES) may present opportunities to increase the perceived value of these resources[296].

Quantitative modeling results presented above relied on numerous assumptions and uncertain parameters (Tables E.6-E.11), but the variability of these outputs was highly sensitive (i.e., having Spearman's coefficients with high absolute values) relative to only a few key groups of parameters (Figure E.1). Perhaps most critical are parameters related to dietary intake and excretion of resources (e.g., phosphorus content of plant-based protein, fraction of energy intake that is excreted), which were highly correlated to potential resource recovery quantities and GHG emissions (as direct methane and N_2O emissions depend on the quantities of COD and nitrogen entering the system).

The above connections reveal the fundamental role of the human-derived resource system – a key feature of the sanitation SES framework that encompasses sanitation users and the resource units they generate (A.S.2.1, A.S.2.2, RU.N.1, RU.O.1) – in affecting multi-dimensional sanitation outcomes (SAN.O). Characterizing the locality-specific attributes of this system (e.g., caloric and protein intake, resource content of intake, and excretion rates into toilet facilities) will be critical in predicting and evaluating sanitation performance and sustainability. Other parameters contributing to the sensitivity of resource recovery potential and GHG emissions include those focused on nutrient leaching from latrines and biological degradation rates, which will depend upon local ecological setting (e.g., soil characteristics, climate; ECO.L.2, ECO.C), as well as the anaerobic baffled reactor's COD removal efficiency in Scenario B. Across the three scenarios, costs are most sensitive to the number of people served by each toilet (determined by household size and use density) and per capita rates of sludge accumulation in latrines. These parameters may be especially important in densely-populated urban areas such as informal settlements, where user numbers may vary widely depending on how many households share a single toilet (A.S.1.1, A.S.3.1, S.S.1.1).

An understanding of local perceptions relative to the feasibility of various sanitation scenarios is necessary to move beyond quantitative comparisons of performance, integrating modeling results with findings from discussions or surveys focused around the perspectives, concerns, and questions of stakeholders. For example, approximately two-thirds of survey respondents expressed that they were satisfied with their current sanitation facilities (pit latrines in most cases), suggesting that a transition toward a container-based system using different onsite facilities may require robust sensitization efforts to ensure buy-in of the community. Additionally, if design and operation of container-based facilities could address current issues with existing latrines, social feasibility may increase. For example, those not satisfied with sanitation facilities reported issues related to the cleanliness of the latrine (66%), long waiting times (25%), and fear for personal safety when using the facilities (24%). Furthermore, a container-based system that alters the conveyance stage such that the role of trucking is reduced or eliminated should consider the effects on truck operators (e.g., job losses). Minimizing the negative impacts of such a transition may necessitate new training to help truck operators find economically beneficial opportunities, perhaps filling roles within the new sanitation economy. For example, they might help to transport recovered nutrient or energy products to markets. In contrast, a shift toward a scenario that only changes the approach to centralized treatment without altering onsite facilities or conveyance (Scenario B) is less likely to affect the sanitation experiences of latrine users or truck operators (although the recovery of biogas could result in new products these groups could purchase and use).

Implications for research and decision-making. Embedding quantitative modeling analyses within a broader conceptual framework can enable researchers and stakeholders to examine, understand, and discuss multi-dimensional impacts of alternative development strategies using a shared vocabulary[271]. Ultimately, actors can identify specific local approaches and generalizable global trends that increase the likelihood of sanitation success and sustainable resource management. For researchers collaborating with local partners to investigate diverse sanitation-related questions, it can help to identify the variables and parameters explicitly included in their models, as well as additional attributes of the system that are closely linked with those primary variables, while also illuminating other system features that may be outside the scope of their work. In our illustrative example, we focused primarily on the resource recovery potential, GHG emissions, and economics associated with alternative sanitation scenarios and explicitly included parameters most needed to characterize those outcomes. However, we also used previous work and experience to consider additional related issues such as soil suitability of recovery products[295], proximity of recovered nutrients to crop demands[202], and ecosystem services[296], while acknowledging that our analysis did not explicitly include other important topics such as health risks. As such, our analysis may provide decision-makers (e.g., government officials, city planners, utilities, non-governmental organizations) with useful insight into local sanitation alternatives (although this work represents an illustrative example that should be expanded upon in the future), and we also have a standardized vocabulary for clarifying limitations and assumptions.

Specifically, the analysis suggests that CIDI's proposed treatment center may provide economic, environmental, and resource management advantages over existing plant designs, while requiring minimal changes to current fecal sludge management infrastructure (pit latrines and truck conveyance, thereby minimizing changes in local perceptions of onsite sanitation facilities). In particular, capturing biogas reduces GHG emissions and resource losses, and it represents a product that may generate revenue and reduce households' cooking fuel expenditures. However, bottling recovered biogas to create a marketable product may present new challenges, while existing issues around access and affordability of pit emptying services will diminish potential benefits if left unaddressed.

Alternatively, container-based sanitation would likely necessitate a more fundamental change to local sanitation approaches (i.e., waterborne systems if possible, with centralized sludge management for populations with pit latrines). Given its dependence on robust, low-cost conveyance[96], a semi-centralized approach to treatment (with small facilities located in or near communities) that is specifically tailored to safely recover marketable products from source-separated inputs may be most effective. Urban areas where sanitation infrastructure is already well-established (representing sunk costs and long-term investments) may exhibit strong physical, institutional, and behavioral inertia[279,316] (S.E.5.2), hindering any transition toward approaches that require new toilet facilities, different (and more reliable) collection services, and potentially new treatment processes. However, where sanitation remains underdeveloped (e.g., some informal settlements), infrastructure investments could bypass conventional approaches and generate movement toward alternative systems that may present more sustainable opportunities. Findings such as these can feed into stakeholder engagement to evaluate scenarios and build consensus, potentially incorporating decision-support tools and visual representations to aid comprehension and system understanding[44,97,302] (SAN.D.1, SAN.D.3).

More generally, this sanitation SES framework begins to provide an overarching, guiding structure to inform the development of models and their application to support decision-making. It represents a mechanism through which multiple models can be integrated or compared with one another, while also revealing contextual aspects that may play a key role in decision-making but are not easily incorporated into quantitative tools. As with the general SES framework's development and application across numerous contexts[265,269–272], employing the sanitation framework to examine various issues across multiple settings and systems can support ongoing work to provide generalizable insight into appropriate strategies for implementing successful sanitation systems that address local priorities[37,256,263]. Furthermore, stimulating diverse use across disciplines can support an iterative process whereby the framework is expanded, operationalized, and improved over time. In the end, we expect the sanitation SES framework will enable interdisciplinary teams to systematically study and make informed decisions about this unique category of human-derived resource systems, driving sustainable development at the interface of nature and society.

CHAPTER 7: CONCLUSIONS AND ENGINEERING SIGNIFICANCE

Development and implementation of innovative sanitation systems designed for resource recovery present considerable possibilities for amplifying sustainable global development, but numerous interconnected issues may constrain whether various approaches are locally viable. In particular, resource-limited settings such as urban informal settlements lacking safe and reliable sanitation offer opportunities to develop innovative systems that capitalize on the potential benefits and amplifying effects of resource recovery more effectively than conventional approaches (e.g., waste sequestration in pit latrines, wastewater treatment via conventional activated sludge). These efforts may address multiple challenges faced by vulnerable populations (e.g., sanitation access, agricultural nutrient availability, household energy, employment opportunities). However, design and implementation processes must ensure that these systems are locally appropriate by interfacing with relevant stakeholders, employing holistic frameworks to understand available options and identify connections with other aspects of sustainability, and establishing quantitative methodologies for assessing alternatives relative to multiple goals and constraints. The modeling approaches and conceptual frameworks presented in this dissertation form a foundation for rigorous, multidimensional analyses that can restrain over-activism for contextually mismatched "one size fits all" technologies. Furthermore, in combination with increasing pressure on planetary boundaries and the growing need to advance circular economies and establish more sustainable resource flows, these efforts begin to provide evidence for overcoming inertia and inspiring an integrated paradigm in which technical design interacts with social, ecological, and economic systems to generate mutually beneficial outcomes.

Key Implications Relative to Each Objective

Estimating the quantities of human-derived nutrients and energy that could be recovered in the future from newly-installed or upgraded sanitation systems at global, regional, and national levels (Objective i; Chapter 2) suggests that resource recovery from sanitation provides an opportunity to simultaneously benefit multiple societal goals, including universal sanitation access, improved availability of agricultural nutrients, and more sustainable household energy. Globally, the nitrogen, phosphorus, and potassium recoverable from sanitation systems installed by 2030 to meet the needs of populations without basic access may offset 5-16% of projected inorganic fertilizer use. The global potential for recoverable energy to offset household electricity demands is an order of magnitude smaller (0-2%), and household electricity represents only a small fraction of total energy consumption. Resource recovery, therefore, can have a substantial impact on global nutrient cycles and play a role in the formation of more sustainable agricultural systems. The diversity of country-specific findings suggests the most effective pathways for future sanitation will likely vary depending on local context. Developing appropriate strategies may be especially impactful among the least-developed countries, six of which could double or offset all projected nutrient and household electricity use through resource recovery from newly-installed sanitation. An initial co-location analysis reveals that many of those countries with the largest potential impacts also display a high degree of spatial alignment between recoverable nutrients and agricultural needs, providing further support for efforts to capitalize on the amplifying effects of resource recovery in resource-limited settings.

Building upon this co-location analysis, examining nutrient recovery potential and transport distance for 56 of the world's largest cities (Objective ii; Chapter 3) offers insight into pathways for closing urban nutrient cycles. The spatial characteristics associated with recirculating recovered nutrients from urban sanitation systems to surrounding cropland reveal where and how certain nutrient recovery processes and products may be constrained by transport distance and energy requirements. Across this set of 56 cities, estimated distances span two orders of magnitude, and transport needs tend to be smaller among European, African, and Asian cities due to factors such as high local cropland density, compact (i.e, less sprawling) urban areas, and local prevalence of nutrient-intensive crop types. In cities associated with longer transport distances, recovery of highly concentrated products (e.g., struvite, ammonium sulfate) may be needed to make distant transport (or export) feasible by reducing transport energy requirements. Strategies capitalizing on nutrient reuse may be particularly impactful for smallholder farmers, many of whom live in Africa and Asia (where distances are often estimated to be shorter), potentially improving economic and food security of vulnerable populations while providing resilience against international fertilizer and food price spikes. However, these findings on the potential linkages between cities and rural cropland should be complemented with place-based studies able to holistically consider the characteristics of specific infrastructure, agricultural, and other locally-relevant systems. For example, although findings from a sensitivity analysis suggest that incorporating road networks when estimating transport distances would not change the broad trends observed in this study, detailed network modeling that considers road location, connectivity, and quality would provide more accurate results when conducting focused analyses within specific cities.

Expanding these global spatial modeling methods to evaluate the soil suitability of recovered nutrient products (Objective iii; Chapter 4) suggests how the agronomic value of each product (reclaimed wastewater, source-separated urine, digested sludge, compost, ammonium sulfate, ammonium struvite, potassium struvite) may depend upon the interactions between product chemistry and soil context. Soil pH tends to be a key driver of suitability for alkaline or acidic products, indicating locations where these products may be beneficial or detrimental to agriculturally conducive soil conditions. For example, struvite (an alkaline product) may be particularly beneficial when applied to acidic soils in Uganda but detrimental to alkaline soils in the southwestern United States. Other parameters, including soil cation exchange capacity and clay content, often determine where products with highly mobile nutrients (e.g., reclaimed wastewater) may have reduced capacity to meet crop demands due to issues surrounding nutrient retention and loss. Integrating this suitability information with the nutrient recovery potential of each product reveals a wide variety of soil suitability and co-location patterns across countries, suggesting possible avenues in which certain countries or regions may focus on recovery of one or more specific products. Alternatively, incorporating considerations of transport distance may enable stakeholders to explore potential export of certain products to locations where soil context may be more favorable. Broadly, incorporating even the coarsest information regarding soil context may markedly improve global assessments concerning the contextual appropriateness of nutrient recovery strategies. However, this analysis also uncovers considerable discrepancies existing across soil datasets that employ differing methodologies, scales, and extents, highlighting the need for locally accurate knowledge and interpretation when considering sanitation alternatives in a given setting.

The preceding global-scale analyses identify opportunities, constraints, and trends relating to a number of factors. Alongside locality-specific issues around broader environmental impacts, economic viability, and social appropriateness, the previous findings emphasize the need for conceptual frameworks that connect sanitation with other relevant systems and assess multidimensional outcomes in specific contexts. One area that has received particularly limited attention involves potential linkages between resource recovery and ecosystem services. Developing a conceptual framework that characterizes the mutually beneficial interactions between these engineered and natural systems (Objective iv; Chapter 5) sheds light on new or alternative pathways for deriving further value from sanitation and resource recovery, particularly in settings with extensive ecological assets but limited economic means. The conceptual framework elucidates numerous options directly and indirectly connecting different types of resources to supporting, regulating, and provisioning ecosystem services. Beyond the framework itself, this work builds on previous methods to evaluate the spatial co-location of recoverable resources and distinct land cover types where certain ecosystem services may be most relevant. While resource recovery can provide households or communities with a considerable degree of freedom regarding how resources are recovered and used, a scenario where each household follows its own path may not generate impactful benefits on a larger system-wide level. In countries where a given land cover type is dominant, for example, cooperation and consensus-building efforts to develop collective sanitation strategies that coordinate recovery and direct resources toward specific ecosystem services may function to aggregate the individual impacts of many households and produce broader-scale change. Additional issues around sanitation technology choice and financing mechanisms are also examined, pointing toward the need for future research that quantitatively estimates the ecological impacts of resource recovery and their potential value to human populations across different contexts. This framework, together with future efforts, can contribute toward integrated system design and decision-making tools that effectively couple engineered and natural systems, balancing societal needs with ecosystem functioning.

Finally, developing a broader social-ecological systems framework that conceptualizes sanitation as a human-derived resource system, and using that framework to support analyses in specific contexts (Objective v; Chapter 6), can enable researchers, stakeholders, and other interdisciplinary actors to examine, understand, and discuss multi-dimensional impacts of alternative development strategies using a shared vocabulary[271], to identify specific local approaches and generalizable global trends that increase the likelihood of sanitation success and sustainable resource management. Specifically in Bwaise, Uganda, the illustrative application of this framework suggests a proposed alternative treatment center may provide economic, environmental, and resource management advantages over existing plant designs, while requiring minimal changes to current fecal sludge management infrastructure (pit latrines, truck emptying and conveyance). Alternatively, container-based sanitation would likely necessitate a more fundamental change to local sanitation approaches (potentially appropriate particularly where infrastructure remains relatively underdeveloped and sunk costs do not present considerable inertia hindering prospective transitions), perhaps incorporating a semi-centralized treatment strategy (with small facilities located in or near communities) specifically tailored to safely recovering marketable products from source-separated inputs. Findings such as these can feed into stakeholder engagement to evaluate scenarios and build consensus, potentially incorporating decision-support tools and visual representations to aid comprehension and system understanding. More generally, this sanitation SES framework begins to provide an overarching, guiding structure to inform the development of models and their application to support decision-making around sanitation. Employing the framework to examine various issues across multiple settings can support ongoing work to provide generalizable insight into appropriate strategies for implementing successful sanitation systems that address local priorities. Furthermore, diverse use can support an iterative process whereby the framework is expanded, operationalized, and improved over time. In the end, we hope the sanitation SES framework will enable interdisciplinary teams to integrate the analyses presented in this dissertation into holistic initiatives that systematically study and make informed decisions about sanitation, amplifying sustainable development at local and global levels.

Limitations and Future Avenues for Research

The studies included in this dissertation represent a foundation for analyzing several issues and opportunities surrounding resource recovery from sanitation, both in specific, single-issue approaches (useful for evaluating a certain consideration across multiple large-scale settings) and multidimensional integrated frameworks (particularly valuable when considering multiple outcomes and tradeoffs in individual contexts). Future work can build on the approaches developed and applied here via multiple pathways, to link distinct models more explicitly, operationalize integrated analyses across contexts, and effectively communicate findings from these analyses to stakeholders and decision-makers at multiple levels. The following paragraphs detail four categories of future research to expand upon the work in this dissertation, address its limitations, and strengthen its impact on multiple aspects of sustainable development.

Refine global analysis methods and input datasets. While the procedures used to estimate sanitation's potential impacts on resource access, spatial co-location and transport requirements, and soil suitability at a global scale (Chapters 2-4) were able to account for many key drivers (e.g., dietary intake, sanitation coverage levels, population density, distinct nutrient demands of various crops), they also relied upon several assumptions and simplifications. Future work could focus on identifying alternative or supplemental input data to address the considerable uncertainties and incomplete nature of some datasets (e.g., food waste, wastewater production and treatment, road networks and other transport infrastructure, soil conditions). Certain geographic locations could be matched with one or more appropriate datasets (potentially with multiple options providing different degrees of spatial resolution, which can be selected based on the desired application or level of detail). These refinements would offer further insight into how improvements in data may influence the findings of specific analyses (see Chapter 4 for an example related to soil data in sub-Saharan Africa) and supply better information for future studies focused in particular areas.

Integrate additional dimensions into frameworks. The conceptual frameworks developed to examine resource recovery from sanitation in connection with ecosystem services and social-ecological systems (Chapters 5-6) offer new ways of exploring sanitation's impacts and interactions with other aspects of sustainable development. In particular, quantitative modeling related to the social-ecological systems framework has established explicit links across economics, broader environmental impacts (greenhouse gas emissions), and resource recovery potential. Moving further, future work can integrate additional dimensions to present a more holistic picture of the potential opportunities and challenges associated with various sanitation systems across diverse contexts. For example, quantitative models can explicitly include health risks for consumers purchasing recovered products and for sanitation workers (e.g., truck operators in Bwaise), connections between agricultural application of recovered nutrient products and increases in crop yields (with resulting economic impacts), and a more developed understanding of existing and potential demand for resources (e.g., fertilizers). In particular, farmers in rural areas and around smaller urban areas (e.g., secondary and tertiary cities) may benefit from increased fertilizer access, but the costs associated with transport of fertilizers to distribution centers outside of primary cities (e.g., Kampala) and from these centers to farmers (i.e., the “last mile” problem) may be prohibitive[317]. Considering the potential for less centralized sanitation systems that could provide resources in a more evenly distributed manner may help to address these transport issues and increase demand higher than might otherwise be seen, while simultaneously improving sanitation in potentially less-developed areas. Populations in these locations may also be more accepting of transformative and potentially disruptive sanitation solutions if existing technologies are lacking or underdeveloped. From a qualitative perspective, applications of the sanitation SES framework will require greater development to more directly incorporate an understanding of governance systems, socio-political structures, and perceptions of local actors. The quantitative modeling presented in Chapter 6 did not explicitly address issues related to the potential costs of disruptive social changes (e.g., job losses, conflicts with urban planning), perceptions associated with resource recovery in different forms (and how those perceptions compare to conventional alternatives), and power dynamics across different groups (e.g., community members, community leaders, government officials). Recent work updating the general SES framework to better account for power relationships could be integrated into the sanitation framework as well[318–322].

In the ecosystem services framework, hypothetical pathways connecting recoverable resources and various services could be made more concrete by including standardized analysis procedures related to spatial co-location with key land cover types and financing mechanisms (see Chapter 5 for further discussion of these issues). Incorporating efforts to quantify the potential impact of recoverable resources on specific ecosystem services (e.g., carbon sequestration through land application of organic matter; carbon credits associated with biogas recovery offsetting fossil energy) would also help to define the ecological value of these pathways in multiple settings. This framework could also be expanded by reversing the directionality of many connections between ecosystems and recoverable resources, to further explore the impacts ecosystems and the services they provide may affect the potential to recover resources (e.g., in natural treatment processes associated with wetlands). Furthermore, the procedures employed in global analyses around transport requirements and soil suitability (Chapters 3-4) could be adapted for more focused, smaller-scale application, such that these considerations can be integrated within models related to the ecosystem services and social-ecological systems frameworks in a more quantitative manner to develop suitability mapping procedures around sanitation and resource recovery that go beyond consideration of a single issue (although context-relevant findings from the global analyses in Chapters 3-4 were discussed in Chapter 6, these spatial analysis procedures were not explicitly integrated into the contextually-defined models developed for Bwaise, Uganda).

Apply these integrated frameworks across various technologies, settings, and scales. After expanding the frameworks to provide insight into additional issues (e.g., spatial co-location, transportation, financing), future work can apply them across multiple settings to develop a generalizable understanding of resource recovery from sanitation and the critical drivers that affect its viability, success, and sustainability. The social-ecological systems framework has been partially applied to consider a limited set of sanitation alternatives in a single context (an urban informal settlement in Kampala, Uganda; Chapter 6), while the ecosystem services framework was developed conceptually to stimulate future focused research into the potential connections between resource recovery and ecosystem services. As such, these frameworks should be operationalized and applied across many technologies and settings at varying scales (e.g., household, community, city, country), using these experiences to broaden future applicability and incorporate alternative or supplemental data sources and collection methods as needed. Critically, diverse applications can identify or better characterize key variables and relationships that tend to drive the sustainability and successful multidimensional performance of conventional and novel sanitation systems, eventually offering generalizable insight for more effective decision-making around system design, implementation, and operation. Furthermore, these applications may provide insight into how interactions across multiple system features can be operationalized. For example, recovery potential relative to different nutrient products should be considered in tandem with an understanding of existing access to and demand for fertilizers, as well as the impacts of increased nutrient application on crop yields. Similarly, sanitation approaches that reduce greenhouse gas emissions may offer opportunities to develop financing strategies related to ecosystem services (e.g., carbon credits).

Develop complementary communication tools for stakeholder engagement. Beyond expanding upon these analytical methods and frameworks, future research can also work to develop effective communication tools able to educate and inform various stakeholders regarding the potential outcomes and impacts of alternative sanitation and development scenarios. These communication tools should be developed in collaboration with local implementation partners to ensure they are contextually appropriate and provide a suitable level of detail. Ideally, they would eventually act as an integrated complement to the quantitative analysis models and methods put forward in this dissertation, perhaps using graphical or pictorial means to convey complex information in an efficient, streamlined, and easily understood manner. In addition to communication tools designed to inform stakeholder understanding and decision-making among local partners, a set of online interfaces and dashboards could be created, making the results of these analyses widely available to planners, funders, implementers, and other interested parties around the world. If the methods behind the analyses could also be embedded into these online tools, then an array of stakeholders could begin to apply these approaches to sanitation and resource management across a range of local settings. By broadening the reach of these analysis methods in this way, they can have a greater impact in supporting the development and implementation of sanitation and resource recovery strategies that protect the environment, promote circular resource economies, encourage mutually beneficial interactions between engineered, natural, and social systems, and amplify sustainable development.

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APPENDIX A: SUMMARY OF SUPPORTING MATERIALS FOR CHAPTER 2

All Supporting Materials related to Chapter 2 are available online at:

https://pubs.acs.org/doi/suppl/10.1021/acs.est.7b02147

This address links to a page on the *Environmental Science & Technology* website containing Supporting Materials for the following article, reprinted in Chapter 2:
Trimmer, J. T.; Cusick, R. D.; Guest, J. S. Amplifying Progress toward Multiple Development Goals through Resource Recovery from Sanitation. *Environ. Sci. Technol.* **2017**, *51* (18), 10765–10776.

Supporting Materials for Chapter 2 are organized as follows:

Supplementary Methods

- Section S1. Data collection and preliminary handling procedures
- Section S2. Projection scenarios to estimate future conditions
- Section S3. Procedures to estimate populations served by sanitation system categories
- Section S4. Procedures to estimate potential impacts of resource recovery
- Section S5. Procedures for aggregating across regions and the world
- Section S6. Spatial co-location analysis
- Section S7. Co-location sensitivity analysis
- Section S8. Overall uncertainty analysis
- Section S9. Procedure to estimate energy recovery for household cooking

Supplementary Figures

- Figure S1. Logic flow diagram for resource recovery impacts
- Figure S2. Diagram showing the flow of data and calculations in this analysis
- Figure S3. Sanitation coverage relative to other development indicators by country
- Figure S4. Summary of impacts estimated from all projection scenarios
- Figure S5. Sensitivity analysis of recovery technology efficiencies
- Figure S6. Regional summaries from each sanitation system category
- Figure S7. Possible replacement of imported nutrients through resource recovery
- Figure S8. Sensitivity analysis of co-location results

Supplementary Tables

- Table S1. The Sustainable Development Goals and links with resource recovery
- Table S2. Summary of datasets used in this analysis and Figure S3
- Table S3. Country-level data from 2010, as presented in Figure S3
- Table S4. Summary of projection scenarios
- Table S5. Parameter values and uncertainty distributions used in this analysis
- Table S6. Summary of datasets considered for the energy recovery impact analysis
- Table S7. Potential impacts from resource recovery by region
- Table S8. Potential impacts from newly-installed sanitation systems by country
- Table S9. Potential impacts from newly-treated wastewater systems by country
- Table S10. Potential impacts from replaced existing systems by country
- Table S11. Recommended nutrient application rates for seventeen major crops
- Table S12. Co-location results and responses to sensitivity analysis by country

APPENDIX B: SUMMARY OF SUPPORTING MATERIALS FOR CHAPTER 3

All Supporting Materials related to Chapter 3 are available online at:

https://www.nature.com/articles/s41893-018-0118-9#Sec19

This address links to a page on the *Nature Sustainability* website containing Supporting Materials for the following article, reprinted in Chapter 3:
Trimmer, J. T.; Guest, J. S. Recirculation of Human-Derived Nutrients from Cities to Agriculture across Six Continents. *Nature Sustainability* **2018**, *1* (8), 427–435.

Supporting Materials for Chapter 3 are organized as follows:

Supplementary Methods
- Supplementary Methods 1. Procedures to estimate recoverable nutrient quantities
- Supplementary Methods 2. Procedures used in the nutrient distance analysis

Supplementary Figures
- Supplementary Figure 1. Defining city extent and conducting the distance analysis
- Supplementary Figure 2. Conceptual summary of the nutrient distance analysis
- Supplementary Figure 3. Example results from the nutrient distance analysis
- Supplementary Figure 4. Average distances from primary and centralized scenarios
- Supplementary Figure 5. Recoverable phosphorus quantities and average distances
- Supplementary Figure 6. Recoverable potassium quantities and average distances
- Supplementary Figure 7. Reductions in average distances from shifts in crop production
- Supplementary Figure 8. Transport energy associated with different recovery products

Supplementary Tables
- Supplementary Table 1. Geographical and typological characteristics of cities
- Supplementary Table 2. Additional characteristics of cities
- Supplementary Table 3. Parameters used to estimate recoverable nutrient quantities
- Supplementary Table 4. Nutrient application rates for nationally significant crops
- Supplementary Table 5. Summary of nutrient distance analysis scenarios
- Supplementary Table 6. Nutrient quantities and distance results (Excel spreadsheet)
- Supplementary Table 7. Spearman correlation analyses (for quantitative factors)
- Supplementary Table 8. Simple linear regression analyses (for quantitative factors)
- Supplementary Table 9. Kruskal-Wallis tests (for categorical factors)
- Supplementary Table 10. Average distances across city groupings (continent, coast)
- Supplementary Table 11. National fertilizer import replacement
- Supplementary Table 12. Parameters used to estimate transport energy of products
- Supplementary Table 13. Comparison of national road lengths from multiple sources
- Supplementary Table 14. Summary of quality control analysis on the impact of roads
- Supplementary Table 15. Complete quality control analysis results (Excel spreadsheet)

APPENDIX C: SUMMARY OF SUPPORTING MATERIALS FOR CHAPTER 4

All Supporting Materials related to Chapter 4 are available online at:

https://pubs.acs.org/doi/suppl/10.1021/acs.est.9b00504

This address links to a page on the *Environmental Science & Technology* website containing Supporting Materials for the following article, reprinted in Chapter 4:
Trimmer, J. T.; Margenot, A. J.; Cusick, R. D.; Guest, J. S. Aligning Product Chemistry and Soil Context for Agronomic Reuse of Human-Derived Resources. *Environ. Sci. Technol.* **2019**, *53* (11), 6501–6510.

Supporting Materials for Chapter 4 are grouped into six themes, as follows:

Theme A. Estimation of global nitrogen flows
- Figure S1. Diagram of global nitrogen flows through agriculture and human populations
- Table S1. Values, ranges, and references for all flows shown in Figure S1
- Table S2. Values, ranges, and references for all outflow categories shown in Figure S1
- Table S3. Summary of calculations used to scale values from disparate years

Theme B. Soil suitability mapping
- Section S1. Descriptions of soil parameters used in this analysis
- Figure S2. Global maps of each soil parameter
- Table S4. Summary of the relevant characteristics of each nutrient recovery product
- Table S5. Matrix showing relationships between soil conditions and each product
- Table S6. Summary of criteria and threshold values used in generating suitability maps
- Table S7. Specific soil types that are more or less prone to phosphorus fixation
- Table S8. Soil suitability classification definitions used in this analysis

Theme C. Nutrient excretion
- Section S2. Methods used to estimate nutrient excretion from human populations
- Table S9. Summary of all parameters used to estimate per capita nutrient excretion

Theme D. Product-specific nutrient recovery
- Section S3. Methods used to estimate nutrient recovery in specific products
- Table S10. Summary of assumptions used to estimate product-specific recovery

Theme E. Soil data sensitivity analysis
- Figure S3. A comparison of soil suitability maps for Africa using two soil datasets

Theme F. Recovery potential, co-location, and suitability results across countries
- Table S11. Nitrogen recovery potential, co-location, and suitability (Excel file)
- Table S12. Phosphorus recovery potential, co-location, and suitability (Excel file)
- Table S13. Potassium recovery potential, co-location, and suitability (Excel file)
- Table S14. Carbon recovery potential, co-location, and suitability (Excel file)

APPENDIX D: SUMMARY OF SUPPORTING MATERIALS FOR CHAPTER 5

All Supporting Materials related to Chapter 5 are available online at:

https://www.nature.com/articles/s41893-019-0313-3#Sec11

This address links to a page on the *Nature Sustainability* website containing Supporting Materials for the following article, reprinted in Chapter 5:
Trimmer, J. T.; Miller, D. C.; Guest, J. S. Resource Recovery from Sanitation to Enhance Ecosystem Services. *Nature Sustainability* **2019**.

Supporting Materials for Chapter 5 are organized as follows:

Supplementary Results. Crossover of sanitation and ecosystem services literature

Supplementary Methods. Estimating resource recovery potential and co-location

Supplementary Figures
- Supplementary Figure 1. Maps of nitrogen recovery potential and dominant land cover
- Supplementary Figure 2. Conceptual figure showing nitrogen recovery from populations

Supplementary Tables
- Supplementary Table 1. Publications on sanitation enhancing ecosystem services
- Supplementary Table 2. Parameters used to estimate recoverable nitrogen
- Supplementary Table 3. Datasets used in the spatial co-location analysis
- Supplementary Table 4. Co-location of resources with dominant land cover by country

APPENDIX E: SUPPORTING MATERIALS FOR CHAPTER 6

All Supporting Materials related to Chapter 6 are included in this appendix.

Supporting Materials for Chapter 6 are grouped into three themes, as follows:

Theme A. Identifying multiple tiers of variables within the sanitation SES framework
- Table E.1. Variables in the *Sanitation* subsystem
- Table E.2. Variables in the *Resource units* and *Reuse systems* subsystems
- Table E.3. Variables in the *Actors* and *Governance systems* subsystem
- Table E.4. Variables in the *Related settings* and *Related ecosystems* subsystems

Theme B. Data collection in Bwaise, Uganda
- Survey E.1. The survey conducted among Bwaise households
- Photographs E.1. Pictures of Bwaise, sanitation systems, partners, and stakeholders

Theme C. Quantitative modeling around sanitation alternatives in Bwaise, Uganda
- Section E.1. General approach to quantitative scenario modeling
- Section E.2. Model inputs and excretion estimates
- Section E.3. Bwaise's existing sanitation system
- Section E.4. CIDI's proposed treatment center
- Section E.5. Container-based sanitation
- Figure E.1. Sensitivity of outputs to key parameters across all scenarios
- Table E.5. Summary of alternative sanitation scenarios
- Table E.6. Parameters used as initial inputs common to all scenarios
- Table E.7. Parameters used in the *User interface* process stage
- Table E.8. Parameters used in the *Decentralized storage* process stage
- Table E.9. Parameters used in the *Conveyance* process stage
- Table E.10. Parameters used in the *Centralized treatment* process stage
- Table E.11. Parameters used in the *Reuse/disposal* process stage

Theme A. Identifying multiple tiers of variables in the Sanitation SES framework

Table E.1. Multiple tiers of variables in the *Sanitation* subsystem.

First tier	Second tier	Third tier	Fourth tier
Sanitation (SAN)	Technology/ system selection (SAN.T)	System design (SAN.T.1)	User interface (SAN.T.1.1)
			Onsite storage/treatment (SAN.T.1.2)
			Conveyance (SAN.T.1.3)
			Centralized treatment/recovery (SAN.T.1.4)
			Reuse/disposal strategies (SAN.T.1.5)
			System scale/density (SAN.T.1.6)
		System operation/maintenance (SAN.T.2)	Expected lifetime (SAN.T.2.1)
			Serviceability/accessibility (SAN.T.2.2)
			Flexibility/adaptability (SAN.T.2.3)
	Management systems (SAN.M)	System operation (SAN.M.1)	User requirements (SAN.M.1.2)
			Storage needs (SAN.M.1.3)
			Collection/transport needs (SAN.M.1.4)
			Treatment/recovery needs (SAN.M.1.5)
			Reuse/disposal needs (SAN.M.1.6)
		Monitoring/evaluation (SAN.M.2)	-
		Financing options/environment (SAN.M.3)	Access to credit (SAN.M.3.1)
			Local markets for products (SAN.M.3.2)
	Decision-making processes (SAN.D)	Stakeholder participation (SAN.D.1)	Engagement mechanisms/intensity (SAN.D.1.1)
			Expert input (SAN.D.1.2)
			Inclusion mechanisms (SAN.D.1.3)
		Development of alternatives (SAN.D.2)	System compatibility (SAN.D.2.1)
			Scenario development (SAN.D.2.2)
		Evaluation and selection (SAN.D.3)	Comprehension aids (SAN.D.3.1)
			Tradeoff/uncertainty analysis (SAN.D.3.2)
			Decision-making tools/models (SAN.D.3.3)
			Cyclic feedback mechanisms (SAN.D.3.4)
			Iterative/incremental processes (SAN.D.3.5)
	Outcomes (SAN.O)	Economic assessment (SAN.O.1)	Capital costs (SAN.O.1.1)
			Operation and maintenance costs (SAN.O.1.2)
			User fees/prices (SAN.O.1.3)
			Resource value/income (SAN.O.1.4)
			Investment potential (SAN.O.1.5)
		Resource balance (SAN.O.2)	Material needs/recovery (SAN.O.2.1)
			Energy needs/recovery (SAN.O.2.2)
			Water needs/recovery (SAN.O.2.3)
			Land requirements (SAN.O.2.4)
			Resource access impacts (SAN.O.2.5)
		Environmental impacts (SAN.O.3)	Water pollution (SAN.O.3.1)
			Air pollution (SAN.O.3.2)
			Soil contamination (SAN.O.3.3)
			Climate change potential (SAN.O.3.4)
			Conservation potential (SAN.O.3.5)
		Human health impacts (SAN.O.4)	Contaminant risk assessment (SAN.O.4.1)
			Nutritional security (SAN.O.4.2)
			Disease prevalence (SAN.O.4.3)
		Social acceptance/impact (SAN.O.5)	Regulatory compatibility (SAN.O.5.1)
			Community/user ownership (SAN.O.5.2)
			Sanitation marketing/diffusion (SAN.O.5.3)
			Ease of use/operation (SAN.O.5.4)
			Employment/skills training (SAN.O.5.5)
			Stakeholder learning (SAN.O.5.6)
			Stakeholder choice/agency (SAN.O.5.7)
			Well-being/equity (SAN.O.5.8)
		Robustness (SAN.O.6)	Technical maturity (SAN.O.6.1)
			Performance uncertainty (SAN.O.6.2)
			Sensitivity to shocks (SAN.O.6.3)
			Sustained use (SAN.O.6.4)

Table E.2. Multiple tiers of variables in the *Resource units* and *Reuse systems* subsystems.

First tier	Second tier	Third tier	Fourth tier
Resource units (RU)	Nutrients (RU.N)	Generation rates (RU.N.1)	-
		Value (RU.N.2)	Possible uses (RU.N.2.1)
		Recovery products (RU.N.3)	Resource density (RU.N.3.1)
			Mobility/transportability (RU.N.3.2)
			Storage feasibility (RU.N.3.3)
		Conventional alternatives (RU.N.4)	Local availability (RU.N.4.1)
	Organic matter (RU.O)	Generation rate (RU.O.1)	-
		Value (RU.O.2)	Possible uses (RU.O.2.1)
			Energetic content (RU.O.2.2)
		Recovery products (RU.O.3)	Resource density (RU.O.3.1)
			Mobility/transportability (RU.O.3.2)
			Storage feasibility (RU.O.3.3)
		Conventional alternatives (RU.O.4)	Local availability (RU.O.4.1)
	Water (RU.W)	Generation rate (RU.W.1)	-
		Value (RU.W.2)	Possible uses (RU.W.2.1)
		Conventional alternatives (RU.W.3)	Local availability (RU.W.3.1)
Reuse systems (RS)	Agriculture (RS.A)	Nutrient needs/use rates (RS.A.1)	Crop patterns (RS.A.1.1)
			Growing seasons (RS.A.1.2)
			Soil nutrient balance (RS.A.1.3)
		Existing nutrient sources (RS.A.2)	Inorganic fertilizer (RS.A.2.1)
			Animal manure (RS.A.2.2)
			Crop/food waste (RS.A.2.3)
		Production impacts (RS.A.3)	Crop yield response (RS.A.3.1)
			Economic gains (RS.A.3.2)
			Environmental effects (RS.A.3.3)
	Energy (RS.E)	Energy needs/use rates (RS.E.1)	Cooking (RS.E.1.1)
			Heating (RS.E.1.2)
			Lighting (RS.E.1.3)
		Existing energy sources (RS.E.2)	Biomass (RS.E.2.1)
			Electricity (RS.E.2.2)
			Fossil fuel products (RS.E.2.3)
		Impacts (RS.E.3)	Economic effects (RS.E.3.1)
			Environmental effects (RS.E.3.2)
		Existing infrastructure (RS.E.4)	Stove type/efficiency (RS.E.4.1)
	Water (RS.W)	Water needs/use rates (RS.W.1)	Drinking (RS.W.1.1)
			Cooking (RS.W.1.2)
			Bathing/cleaning (RS.W.1.3)
			Irrigation (RS.W.1.4)
		Existing water sources (RS.W.2)	Groundwater (RS.W.2.1)
			Surface water (RS.W.2.2)
			Rainwater (RS.W.2.3)
		Impacts (RS.W.3)	Economic effects (RS.W.3.1)
			Environmental effects (RS.W.3.2)
	General (RS.G)	Demand/supply proximity (RS.G.1)	-
		Demand variations (RS.G.2)	Temporal variations (RS.G.2.1)
			Spatial variations (RS.G.2.2)
		Storage capacity (RS.G.3)	-
		Transport capacity (RS.G.4)	-
		System boundaries (RS.G.5)	-
		Reuse infrastructure (RS.G.6)	Acquisition (RS.G.6.1)
			Maintenance (TS.G.6.2)

Table E.3. Multiple tiers of variables in the *Actors* and *Governance systems* subsystems.

First tier	Second tier	Third tier	Fourth tier
Actors (A)	Sanitation users (A.S)	Distribution (A.S.1)	Population/household density (A.S.1.1)
			Demographics (A.S.1.2)
			Socioeconomic status (A.S.1.3)
		Diet (A.S.2)	Caloric intake (A.S.2.1)
			Plant and animal protein intake (A.S.2.2)
		Existing sanitation (A.S.3)	System type and access level (A.S.3.1)
			Current sanitation behaviors (A.S.3.2)
		Preferences/norms (A.S.4)	Historical preferences/behaviors (A.S.4.1)
			Cultural norms/beliefs (A.S.4.2)
			Gender roles/norms (A.S.4.3)
		Decision drivers (A.S.5)	-
	Resource users (A.R)	Distribution (A.R.1)	Population/household density (A.R.1.1)
			Demographics (A.R.1.2)
			Socioeconomic status (A.R.1.3)
		Proximity to sanitation resources (A.R.2)	-
		Existing resource access (A.R.3)	-
		Preferences/norms (A.R.4)	Historical preferences/behaviors (A.R.4.1)
			Cultural norms/beliefs (A.R.4.2)
			Gender roles/norms (A.R.4.3)
		Decision drivers (A.R.5)	-
	Technology operators (A.T)	Distribution (A.T.1)	Population/household density (A.T.1.1)
			Demographics (A.T.1.2)
			Socioeconomic status (A.T.1.3)
		Skills/knowledge (A.T.2)	Education/experience level (A.T.2.1)
			Labor availability/flexibility (A.T.2.2)
	Utilities and businesses (A.U)	Knowledge/experience level (A.U.1)	Technology expertise (A.U.1.1)
			Capacity to innovate (A.U.1.2)
		Decision drivers (A.U.2)	-
	Social capital (A.C)	Trust/reciprocity (A.C.1)	-
		Social network structure (A.C.2)	Degree (A.C.2.1)
			Strength (A.C.2.2)
			Clustering (A.C.2.3)
			Centrality (A.C.2.4)
Governance systems (GS)	Government authorities (GS.G)	Governance frameworks (GS.G.1)	Decision-making processes (GS.G.1.1)
			Procedures for creating rules (GS.G.1.2)
			Property/ownership rights systems (GS.G.1.3)
			Standards/regulations (GS.G.1.4)
			Monitoring/enforcement (GS.G.1.5)
		Scale/boundaries of governance (GS.G.2)	-
		Knowledge sharing processes (GS.G.3)	-
	Regulating agencies (GS.R)	Governance frameworks (GS.R.1)	Decision-making processes (GS.R.1.1)
			Standards/regulations (GS.R.1.2)
			Monitoring/enforcement (GS.R.1.3)
		Scale/boundaries of governance (GS.R.2)	-
	NGOs/civil society (GS.N)	Governance frameworks (GS.N.1)	Decision-making processes (GS.N.1.1)
			Standards/regulations (GS.N.1.2)
			Monitoring/accountability (GS.N.1.3)
		Scale/boundaries of governance (GS.R.2)	-
		Knowledge sharing processes (GS.R.3)	-
	Communities (GS.C)	Governance frameworks (GS.C.1)	Decision-making processes (GS.R.1.1)
			Property/ownership rights systems (GS.R.1.2)
		Community structure/size (GS.C.2)	-
		Social network structure (GS.C.3)	-

Table E.4. Multiple tiers of variables in *Related settings* and *Related ecosystems* subsystems.

First tier	Second tier	Third tier	Fourth tier
Related settings (S)	Social (S.S)	Demographic trends (S.S.1)	Population (S.S.1.1)
			Urbanization (S.S.1.2)
			Migration (S.S.1.3)
			Gender (S.S.1.4)
			Age (S.S.1.5)
		Cultural norms (S.S.2)	Group identity/cohesion (S.S.2.1)
			Shared beliefs/perceptions (S.S.2.2)
			Sacred/protected values (S.S.2.3)
		Human capital/institutions (S.S.3)	Education (S.S.3.1)
			Health status (S.S.3.2)
			Social safety nets (S.S.3.3)
	Economic (S.E)	Level of development (S.E.1)	-
		Income distribution (S.E.2)	-
		International environment (S.E.3)	Trade (S.E.3.1)
			Foreign aid/investment (S.E.3.2)
		Financial markets (S.E.4)	Interest/discount rates (S.E.4.1)
			Exchange rate volatility (S.E.4.2)
			Tax rates (S.E.4.3)
			Access to savings and credit (S.E.4.4)
		Resource markets and incentives (S.E.5)	Technology/material availability (S.E.5.1)
			Infrastructure access/quality (S.E.5.2)
			Resource prices (S.E.5.3)
			Supply/value chains (S.E.5.4)
			Production/consumption patterns (S.E.5.5)
	Political (S.P)	Type(s) of government (S.P.1)	Corruption/good governance (S.P.1.1)
			Media environment (S.P.1.2)
		National/regional stability (S.P.2)	International relations (S.P.2.1)
			Refugee environment (S.P.2.2)
Related Ecosystems (ECO)	Land (ECO.L)	Land cover/use (ECO.L.1)	Temporal changes (ECO.L.1.1)
			Spatial heterogeneity (ECO.L.1.2)
		Soil characteristics (ECO.L.2)	pH (ECO.L.2.1)
			Texture (ECO.L.2.2)
			Salinity (ECO.L.2.3)
			Organic carbon (ECO.L.2.4)
			Nutrient balances (ECO.L.2.5)
		Topography (ECO.L.3)	Slope (ECO.L.3.1)
		Pathogen contamination (ECO.L.4)	-
	Air (ECO.A)	Greenhouse gas emissions (ECO.A.1)	Carbon dioxide (ECO.A.1.1)
			Methane (ECO.A.1.2)
			Nitrous oxide (ECO.A.1.3)
		Indoor air quality (ECO.A.2)	Particulate matter (ECO.A.2.1)
			Carbon monoxide (ECO.A.2.2)
			Nitrogen and sulfur oxides (ECO.A.2.3)
	Water (ECO.W)	Quality (ECO.W.1)	Biological contamination (ECO.W.1.1)
			Organic matter (ECO.W.1.2)
			Nutrients/eutrophication (ECO.W.1.3)
			Heavy metals (ECO.W.1.4)
			Emerging contaminants (ECO.W.1.5)
		Quantity (ECO.W.2)	Water withdrawal/consumption (ECO.W.2.1)
			Flow/recharge rates (ECO.W.2.2)
			Water stress/scarcity (ECO.W.2.3)
			Groundwater table depth (ECO.W.2.4)
	Climate (ECO.C)	Temperature patterns (ECO.C.1)	-
		Rainfall patterns (ECO.C.2)	-
		Solar radiation (ECO.C.3)	-
		Extreme weather events (ECO.C.4)	-
	Ecosystem services (ECO.ES)	Supporting services (ECO.ES.1)	-
		Regulating services (ECO.ES.2)	-
		Provisioning services (ECO.ES.3)	-
		Cultural services (ECO.ES.4)	-
	Biodiversity (ECO.B)	-	-

Theme B. Data collection in Bwaise, Uganda

Survey E.1. The full household survey conducted among Bwaise households appears below.

Household Survey

PART 1: BASIC INFORMATION AND DEMOGRAPHICS

Q.1 Select your name

 1 Interviewer 1
 2 Interviewer 2
 3 Other: _____________

Q.2 Enter the day of the date (i.e. 22 for March 22)

____

Q.3 Explain the study and ask for informed consent to do a brief interview, obtain a household water sample, household soil sample, and hand wash samples from caregiver or child under 5.

 1 Informed consent obtained
 2 No informed consent
 3 Female head of household or primary caregiver not available

[S - IF THE ANSWER IS 2-3, THEN END SURVEY]

Q.4 Observe the respondent's gender

 1 Female
 2 Male

Q.5 Enter the assigned unique cluster ID:

Cluster ID: ________

Q.6 Enter the assigned unique household ID:

Household ID: ________

Q.7 How long have you lived in this settlement?

Time in Settlement: ________ years (or specify other units)
 1 Don't know

Q.8 Where did you live before moving here?

District/country of origin: _________
□ 1 Don't know

Q.9 (If country of origin is not Uganda) Do you have any financial assets in your country of origin, or do you engage in financial transactions with people in your country of origin?
□ 1 Yes
□ 2 No
□ 1 Don't know

Q.10 Are you married?

□ 1 Yes, married
□ 2 Yes, married, spouse not at settlement
□ 3 No, single (never married)
□ 4 No, single (separated by choice, widowed, divorced)
□ 5 Living with unmarried partner
□ 6 Other

Q.11 Please provide the following information for each person living in your household:

FIRST NAME	**GENDER**	**AGE**	**RELATIONSHIP TO RESPONDENT**	**HAS HIS/HER OWN CELL PHONE**			**HAS USED CELL PHONE FOR FINANCIAL TRANSACTIONS**		
				Yes	No	Don't know	Yes	No	Don't know
			(self)	□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
				□ 1	□ 2	□ 3	□ 1	□ 2	□ 3

Q.12 What is the highest level of education that you have had? (mother)

- ☐ 1 No formal schooling
- ☐ 2 Some primary education
- ☐ 3 Completed primary education
- ☐ 4 Some secondary school
- ☐ 5 Completed secondary school
- ☐ 6 Some college
- ☐ 7 Completed college/university
- ☐ 8 Don't know

Q.13 What is your current occupation?

Current occupation: ________

Q.14 What was your previous occupation before you moved here?

Previous occupation: ________

Q.15 (if answered "yes" or "living with unmarried partner" to Q.10) What is the highest level of education your spouse/partner has had? (father)

- ☐ 1 No formal schooling
- ☐ 2 Some primary education
- ☐ 3 Completed primary education
- ☐ 4 Some secondary education
- ☐ 5 Completed secondary education
- ☐ 6 Some college
- ☐ 7 Completed college/university
- ☐ 8 Don't know
- ☐ 9 Not married

Q.16 (if answered yes to Q.10) What is your spouse's or partner's current occupation?

Spouse's current occupation: ________

Q.17 (if answered yes to Q.10) What was your spouse's or partner's previous occupation before you moved here?

Spouse's previous occupation: ________

PART 2: CHILD HEALTH

Q.18 ***Do not prompt.*** Now I would like to ask you about health issues of children in your neighborhood. What illness do you think that children under the age of 5 in the neighborhood suffer from most often? Select option only.

- ☐ 1 Diarrhea
- ☐ 2 Common cold/fever
- ☐ 3 Pneumonia/respiratory disease
- ☐ 4 Malaria
- ☐ 5 Dengue
- ☐ 6 Malnutrition
- ☐ 7 Skin disease
- ☐ 8 Don't know
- ☐ 9 Other: _______________

Q.19 ***Do NOT offer answers, except 'Anything else?' Check all that are mentioned.*** One illness that some young children get is diarrhea. In your opinion, what causes diarrhea among young children in your neighborhood?

- ☐ 1 Nothing, children do not get diarrhea
- ☐ 2 Contaminated water
- ☐ 3 Contaminated food
- ☐ 4 Contact with sewage/feces
- ☐ 5 Weather
- ☐ 6 Don't know
- ☐ 7 Other: ____________

Q.20 ***Please ask illness symptoms for all children under 5 in household. Start with youngest child.*** How old is the youngest child? Be sure to specify if the answer is in months or years.

Age: __________ (month or years, choose unit option)

Q.21 Can you tell me if your child has had the following health problems recently? ***Prompt on each symptom, tick if yes:

	today	past 2 days	past 7 days	no	don't know
diarrhea	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
3 or more bowel movements in 24 hours	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
watery or soft stool	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
blood in stool	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5

Q.22 Have you observed your youngest child putting soil, mud, clay, or sand into his or her mouth in the past 7 days?

☐ 1 Yes
☐ 2 No, other than normal hand to mouth contact
☐ 3 No, not at all
☐ 4 Don't know

Q.23 (if answered yes for Q.19) How many times in the past 7 days did you observe your child putting soil, mud, clay, or sand into his or her mouth?

☐ 1 Less than once per day
☐ 2 Once per day
☐ 3 Twice per day
☐ 4 Three times per day
☐ 5 Four or more times per day
☐ 6 Don't know

Q.24 (if answered yes for Q.19) About how much soil, mud, clay, or sand did you watch your child put into his or her mouth each time?

☐ 1 Amount of dirt normally on fingers
☐ 2 Amount he/she could hold between two fingers
☐ 3 Half of a handful
☐ 4 A handful
☐ 5 More than a handful
☐ 6 Don't know
☐ 7 Other: __________

Q.25 Is there another child under five in the household?

☐ 1 Yes
☐ 2 No

Q.26-Q.31: if applicable, repeat Q.20-Q.24 for child 2 and ask Q.25 to check for another child
Q.32-Q.37: if applicable, repeat Q.20-Q.24 for child 3 and ask Q.25 to check for another child
Q.38-Q.42: if applicable, repeat Q.20-Q.24 for child 4

PART 3: WATER

Q.43 What is the current *main* source of drinking water for members of your household? *(JMP classification in parentheses)* *** Do not read the answers. Select one option only, and enter any identifying information provided by the respondent.

- ☐ 1 Piped water (*improved*)
- ☐ 2 Public tap/standpipe (*improved*)
- ☐ 3 Tube well/borehole (*improved*)
- ☐ 4 Protected dug well (*improved*)
- ☐ 5 Protected spring (*improved*)
- ☐ 6 Rain water collection (*improved*)
- ☐ 7 Unprotected spring (*unimproved*)
- ☐ 8 Unprotected dug well (*unimproved*)
- ☐ 9 Small water vendor (*unimproved*)
- ☐ 10 Tanker truck (*unimproved*)
- ☐ 11 Bottled water (*improved IF used by choice rather than obligation*)
- ☐ 12 Surface water (e.g., river, pond) (*unimproved*)
- ☐ 96 Other: __________
- ☐ 98 Don't know

Identifying information: _______________________

Q.44 How long does it take to go there, get water, and come back? (enter time in minutes)

Time in minutes: __________
☐ 1 Don't know

Q.45 How much water do you collect for your household each day from this source? (enter number of jerry cans)

Number of jerry cans: __________
☐ 1 Don't know

Q.46 Who is primarily responsible for water collection?

- ☐ 1 Women
- ☐ 2 Men
- ☐ 3 Children
- ☐ 4 Other
- ☐ 5 Don't know

Q.47 ***Please take a picture of the water collection containers lined up.***

Q.48 Are you satisfied with the water supply?

- ☐ 1 Yes
- ☐ 2 No
- ☐ 3 Partially
- ☐ 4 Don't know

Q.49 What is the *main* reason you are not satisfied with the water supply?

☐ 1 Not enough
☐ 2 Long waiting queue
☐ 3 Long distance
☐ 4 Irregular supply
☐ 5 Bad taste
☐ 6 Water too warm
☐ 7 Bad quality
☐ 8 Have to pay
☐ 9 Personal safety
☐ 96 Other: __________
☐ 98 Don't know

Q.50 How safe do you think it is to drink water directly from your main drinking water source?

☐ 1 Very safe
☐ 2 Somewhat safe
☐ 3 Unsafe
☐ 4 Don't know

Q.51 Besides your main drinking water source, does your household have a secondary source for drinking or other purposes, now or during other times of the year? If so, what kind?

☐ 0 Do not use a secondary water source
☐ 1 Piped water (*improved*)
☐ 2 Public tap/standpipe (*improved*)
☐ 3 Tube well/borehole (*improved*)
☐ 4 Protected dug well (*improved*)
☐ 5 Protected spring (*improved*)
☐ 6 Rain water collection (*improved*)
☐ 7 Unprotected spring (*unimproved*)
☐ 8 Unprotected dug well (*unimproved*)
☐ 9 Small water vendor (*unimproved*)
☐ 10 Tanker truck (*improved IF chlorinated*)
☐ 11 Bottled water (*improved IF used by choice rather than obligation*)
☐ 12 Surface water (e.g., river, pond) (*unimproved*)
☐ 96 Other: __________
☐ 98 Don't know

Q.52 How long does it take to go there, get water, and come back? (enter time in minutes)

Time in minutes: __________
☐ 1 Don't know

Q.53 How much water do you collect for your household each day from this source? (enter number of jerry cans)

Number of jerry cans: __________
☐ 1 Don't know

Q.54 ***Please take a picture of the water collection containers lined up.***

Q.55 Can you tell me if you use any of the following methods to clean/treat your drinking water? ***Prompt to ask if the household uses each "always, often, sometimes, or never"***

	Always	Often	Sometimes	Never	Don't know
Boil	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
liquid chlorine	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
chlorine tablets	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Filter - ceramic	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Filter - cloth	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Filter - biosand	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Other: ________	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5

Q.58 Who is primarily responsible for water treatment?

☐ 1 Women
☐ 2 Men
☐ 3 Children
☐ 4 Other
☐ 5 Don't know

Q.59 Please show me where you store your drinking water. *** By observation, are the drinking water containers covered or narrow necked? Note that this is only for drinking water and not water used for other purposes.

☐ 1 All are
☐ 2 Some are
☐ 3 None are
☐ 4 Observation not possible

Q.60 ***Please take a picture of your drinking water storage container.***

Q.61 ***Ask caregiver to prepare a cup of water as they would for a child or for themselves*** Observe how they retrieve the water.

☐ 1 Pouring
☐ 2 Using a cup or utensil to dip into the water
☐ 3 From a spout at the bottom of the container
☐ 4 Other: ____________
☐ 5 Observation not possible

Q.62 Is your household currently participating in any water, sanitation, and hygiene intervention trial or study conducted by any other organization or research team?

☐ 1 Yes
☐ 2 No
☐ 3 Don't know

PART 4: SANITATION

Q.63 What kind of toilet facility does this household use? *(JMP classification in parentheses)* *** Do not read the answers. Select one option only. Note that it is what the household *uses* rather than what they have.

- 3 Pour-flush to pit (*improved*)
- 4 VIP/simple pit latrine with floor/slab (*improved*)
- 5 Composting/dry latrine (*improved*)
- 7 Pit latrine without floor/slab (*unimproved*)
- 8 Service or bucket latrine (*unimproved*)
- 9 Hanging toilet/latrine (*unimproved*)
- 10 No facility, field, bush, plastic bag (*unimproved*)
- 11 Other: _______
- 12 Don't know

Q.64 How many households share this toilet? ***Number of households (including the surveyed household)

- 1 Not shared (1 HH)
- 2 Shared family (2 HH)
- 3 Communal toilet (3 HH or more)
- 4 Public toilet (in market or clinic, etc.)
- 5 Don't know

Q.65 How often is the toilet cleaned?

- 1 At least once a day
- 2 2-3 times a week
- 3 Once a week
- 4 Less than once a week
- 5 Almost never
- 6 Don't know

Q.66 Who is primarily responsible for cleaning the toilet?

- 1 Women
- 2 Men
- 3 Children
- 4 Toilet is not cleaned
- 5 Other
- 6 Don't know

Q.67 Are you satisfied with the toilet facility?

- 1 Yes
- 2 No
- 3 Partially

□ 4 Don't know

Q.68 What is the *main* reason you are not satisfied with the toilet facility?

□ 1 Lack of privacy
□ 2 Long waiting queue
□ 3 Long distance
□ 4 Unclean
□ 5 Inadequate protection from the weather
□ 6 Not enough light
□ 96 Other: __________
□ 98 Don't know

Q.69 How are excreta managed when the toilet facility is full?

□ 1 The pit is filled in, and a new facility is built in a new location
□ 2 The pit is left open, and a new facility is built in a new location
□ 3 The pit is emptied, and the contents are disposed of in a closed pit or are treated
□ 4 The pit is emptied, and the contents are dumped on open land or water
□ 5 Other
□ 6 Don't know

Q.70 (if answered that the pit is emptied for Q.69) Who is primarily responsible for emptying the pit?

□ 1 Women
□ 2 Men
□ 3 Children
□ 4 Other
□ 5 Don't know

Q.71 (if answered that the pit is emptied for Q.69) How often is the pit emptied?

□ 1 Once a year or less
□ 2 Every 2-3 years
□ 2 Every 4 or more years
□ 3 Whenever it is full
□ 4 Other
□ 5 Don't know

Q.72 How long does it take for the toilet facility to fill?

Time: __________ years
□ 1 Don't know

Q.73 How deep is your latrine pit?

Depth: __________ meters
□ 1 Don't know

Q.74 (if yes to having children under 5) The last time [NAME OF YOUNGEST CHILD] passed stools, what was done to dispose of the stools? *** Do not read the answers. Select one option only. Use the name so that the respondent is as specific as possible.

□ 1 Child used toilet/latrine
□ 2 Put/rinsed into toilet or latrine
□ 3 Buried
□ 4 Thrown into garbage
□ 5 Put/rinsed into drain or ditch
□ 6 Left in the open
□ 7 Other: __________
□ 8 Don't know

Q.75 (if chose "thrown into garbage" for previous question) How is household garbage disposed of?

□ 1 Garbage dump
□ 2 In the river
□ 3 On the road
□ 4 In drainage ditch
□ 5 Garbage disposal services
□ 6 No designated place/all over
□ 7 Along railway line
□ 8 Burning
□ 9 Other: _____________
□ 10 Don't know

Q.76 Who is primarily responsible for garbage disposal?

□ 1 Women
□ 2 Men
□ 3 Children
□ 4 Other
□ 5 Don't know

Q.77 How often do you wash your hands?

□ 1 More than three times per day
□ 2 Two to three times per day
□ 3 Once per day
□ 4 Once every 2-3 days
□ 5 Less than once per week
□ 6 Almost never
□ 7 Don't know

Q.78 At what times do you usually wash your hands?

	Always	Sometimes	Rarely	Never	Don't know
After using the toilet	□ 1	□ 2	□ 3	□ 4	□ 5

After cleaning child feces	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Before preparing food	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Before eating	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Before feeding child	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5
Other: ___________	☐ 1	☐ 2	☐ 3	☐ 4	☐ 5

Q.79 ***Please take a picture of the place where the respondent washes his/her hands***
Observe the presence of the following.

☐ 1 Water is available
☐ 2 Soap or detergent is present
☐ 3 Ash, mud, sand is present
☐ 4 Observation not possible

Q.80 Does rain ever cause flooding of your compound and/or household?
☐ 1 Yes, flooding in nearby areas and household
☐ 2 Yes, flooding in nearby areas only
☐ 3 Yes, flooding in household only
☐ 4 No, never
☐ 5 Don't know
☐ 6 Other: _________

Q.81 (if answered yes to Q.80) When was the last time your nearby area or household flooded?
☐ 1 Less than 1 week ago
☐ 2 Less than 1 month ago
☐ 3 Within the last year
☐ 4 More than a year ago
☐ 6 Don't know
☐ 7 Other: _________

PART 5: ASSETS

Q.82 For the purposes of research, would you please tell us your total monthly household income in Ugandan shillings including wages, salaries, profits from sales, rent, etc.? (Enter in UGX per month if possible, otherwise specify per day)

UGX per month: ___________
☐ 1 Don't know

Q.83 What material is the household floor made out of?
☐ 1 Earth
☐ 2 Cement/concrete
☐ 3 Wood
☐ 4 Vinyl
☐ 5 Don't know
☐ 6 Other: ________

Q.84 Does your household have?

	Yes	No	Don’t know
Electricity	□ 1	□ 2	□ 3
Radio	□ 1	□ 2	□ 3
Television	□ 1	□ 2	□ 3
Non-mobile phone	□ 1	□ 2	□ 3
Computer	□ 1	□ 2	□ 3
Refrigerator	□ 1	□ 2	□ 3
Bed	□ 1	□ 2	□ 3
Sofa	□ 1	□ 2	□ 3
Wardrobe (wooden/steel)	□ 1	□ 2	□ 3
Table	□ 1	□ 2	□ 3
Chair	□ 1	□ 2	□ 3

Q.85 Does any member of this household own:

	Yes	No	Don’t know
Watch	□ 1	□ 2	□ 3
Bicycle	□ 1	□ 2	□ 3
Motorcycle or motor scooter	□ 1	□ 2	□ 3
Animal-drawn cart	□ 1	□ 2	□ 3
Car or truck	□ 1	□ 2	□ 3
Boat with a motor	□ 1	□ 2	□ 3
Sewing machine	□ 1	□ 2	□ 3
Clock	□ 1	□ 2	□ 3
Water pump	□ 1	□ 2	□ 3
Grain grinder	□ 1	□ 2	□ 3
Fan	□ 1	□ 2	□ 3

Q.86 Does your family have any of the following types of livestock?

	Yes	No	Don’t know	Number owned	Daytime location			Nighttime location		
					Inside house	Outside house	Don’t know	Inside house	Outside house	Don’t know
Cattle	□ 1	□ 2	□ 3		□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
Goats	□ 1	□ 2	□ 3		□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
Pig	□ 1	□ 2	□ 3		□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
Chicken	□ 1	□ 2	□ 3		□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
Lamb	□ 1	□ 2	□ 3		□ 1	□ 2	□ 3	□ 1	□ 2	□ 3
Other: ________	□ 1	□ 2	□ 3		□ 1	□ 2	□ 3	□ 1	□ 2	□ 3

Q.87 ***Please take a picture of the household’s cooking stove.***

Q.88 ***Do not prompt.*** What type of fuel does your household use for cooking?

☐ 1 Wood
☐ 2 Charcoal
☐ 3 Propane gas
☐ 4 Liquid petroleum gas (LPG)
☐ 5 Other: __________
☐ 6 Don't know

Q.89 How long does it take to go there, collect firewood, and carry it back to your house? (if wood is used as the household's cooking fuel)

Total time: _____________ minutes
☐ 1 Don't know

Q.90 Who is primarily responsible for firewood collection?

☐ 1 Women
☐ 2 Men
☐ 3 Children
☐ 4 Other
☐ 5 Don't know

Q.91 ***Do not prompt.*** How often does someone in your household collect firewood? (if wood is the household's cooking fuel)

☐ 1 More than once per day
☐ 2 Once per day
☐ 3 Once every 2-3 days
☐ 4 Once per week
☐ 5 Less than once per week
☐ 6 Don't know

Q.92 How much does your household spend on cooking fuel every month? (If the household does not collect its own firewood)

UGX per month: __________
☐ 1 Don't know

PART 6: AGRICULTURE

Q.93 Does your household have access to agricultural land for your own cultivation?

☐ 1 Yes
☐ 2 No
☐ 3 Don't know

Q.94 (if answered yes to Q.93) Who is primarily responsible for farming?

☐ 1 Women
☐ 2 Men

☐ 3 Children
☐ 4 Other
☐ 5 Don't know

Q.95 (if answered yes to Q.93) Where is your farmland located? Check all that apply.

☐ 1 Immediately next to the household (on their plot)
☐ 2 In designated agricultural space
☐ 3 Other: ___________
☐ 4 Don't know

Q.96 (if answered "in designated agricultural space" for Q.94) How large is your agricultural land (not immediately next to your household)?

Agricultural area: _________ acres (or specify other units)
☐ 1 Don't know

Q.97 (if answered yes to Q.93) What crops do you grow?

	CROPS GROWN		**FERTILIZER**		**MANURE**	**% SOLD**
	Yes	Approx. size (acres)	Yes	Fertilizer type(s)	Yes	
Maize	☐ 1		☐ 1		☐ 1	
Matooke	☐ 1		☐ 1		☐ 1	
Sweet potatoes	☐ 1		☐ 1		☐ 1	
Irish potatoes	☐ 1		☐ 1		☐ 1	
Wheat	☐ 1		☐ 1		☐ 1	
Rice	☐ 1		☐ 1		☐ 1	
Soybean	☐ 1		☐ 1		☐ 1	
Cassava	☐ 1		☐ 1		☐ 1	
Sorghum	☐ 1		☐ 1		☐ 1	
Groundnut	☐ 1		☐ 1		☐ 1	
Cowpea	☐ 1		☐ 1		☐ 1	
Beans	☐ 1		☐ 1		☐ 1	
Peas	☐ 1		☐ 1		☐ 1	
Yam	☐ 1		☐ 1		☐ 1	
Millet	☐ 1		☐ 1		☐ 1	
Green peppers	☐ 1		☐ 1		☐ 1	
Onions	☐ 1		☐ 1		☐ 1	
Tomatoes	☐ 1		☐ 1		☐ 1	
Eggplant	☐ 1		☐ 1		☐ 1	
Carrots	☐ 1		☐ 1		☐ 1	
Cabbage	☐ 1		☐ 1		☐ 1	
Dodo	☐ 1		☐ 1		☐ 1	
Okra	☐ 1		☐ 1		☐ 1	
Sweet bananas	☐ 1		☐ 1		☐ 1	

Mangoes	☐ 1		☐ 1		☐ 1	
Oranges	☐ 1		☐ 1		☐ 1	
Pineapple	☐ 1		☐ 1		☐ 1	
Passion fruit	☐ 1		☐ 1		☐ 1	
Papaya (pawpaw)	☐ 1		☐ 1		☐ 1	
Other: __________	☐ 1		☐ 1		☐ 1	

Q.98 (if answered yes for fertilizer) Can you estimate how much fertilizer your household bought/received last year?

Fertilizer quantity: ________ kg (or specify other units)
☐ 1 Don't know

Q.99 (if answered yes for fertilizer) Did you use all of the fertilizer that you bought/received? If not, how much was not used?

	Yes	No	Don't know	Quantity/percentage remaining:
All fertilizer used?	☐ 1	☐ 2	☐ 3	

Q.100 (if answered yes for fertilizer) How do you apply fertilizers on your crops?

	Yes	No	Don't know	Comments/details
Pouring on top of soil	☐ 1	☐ 2	☐ 3	
Working into the soil	☐ 1	☐ 2	☐ 3	
Other: __________	☐ 1	☐ 2	☐ 3	

Q.101 (if answered yes to agriculture) How many years has your household been farming in this place?

Length of time: ________ years
☐ 1 Don't know

Q.102 Did your household farm before you moved here?
☐ 1 Yes
☐ 2 No
☐ 2 Don't know

Q.103 (if answered yes to Q.102) What crops did you grow before you came here?

	CROPS GROWN		**FERTILIZER**		**MANURE**	**% SOLD**
	Yes	Approx. size (acres)	Yes	Fertilizer type(s)	Yes	
Maize	☐ 1		☐ 1		☐ 1	

Matooke	☐ 1		☐ 1		☐ 1	
Sweet potatoes	☐ 1		☐ 1		☐ 1	
Irish potatoes	☐ 1		☐ 1		☐ 1	
Wheat	☐ 1		☐ 1		☐ 1	
Rice	☐ 1		☐ 1		☐ 1	
Soybean	☐ 1		☐ 1		☐ 1	
Cassava	☐ 1		☐ 1		☐ 1	
Sorghum	☐ 1		☐ 1		☐ 1	
Groundnut	☐ 1		☐ 1		☐ 1	
Cowpea	☐ 1		☐ 1		☐ 1	
Beans	☐ 1		☐ 1		☐ 1	
Peas	☐ 1		☐ 1		☐ 1	
Yam	☐ 1		☐ 1		☐ 1	
Millet	☐ 1		☐ 1		☐ 1	
Green peppers	☐ 1		☐ 1		☐ 1	
Onions	☐ 1		☐ 1		☐ 1	
Tomatoes	☐ 1		☐ 1		☐ 1	
Eggplant	☐ 1		☐ 1		☐ 1	
Carrots	☐ 1		☐ 1		☐ 1	
Cabbage	☐ 1		☐ 1		☐ 1	
Dodo	☐ 1		☐ 1		☐ 1	
Okra	☐ 1		☐ 1		☐ 1	
Sweet bananas	☐ 1		☐ 1		☐ 1	
Mangoes	☐ 1		☐ 1		☐ 1	
Oranges	☐ 1		☐ 1		☐ 1	
Pineapple	☐ 1		☐ 1		☐ 1	
Passion fruit	☐ 1		☐ 1		☐ 1	
Papaya (pawpaw)	☐ 1		☐ 1		☐ 1	
Other: __________	☐ 1		☐ 1		☐ 1	

Q.104 (if answered yes to Q.100) How many years did your household farm before you moved here?

Length of time: ________ years
☐ 1 Don't know

PART 7: DIET

Q.105 On a typical day, what percentage of your household's diet comes from the following sources?

Provided free of charge (e.g., from NGO, gov.) __________%
Purchased (e.g., from a local market): __________%
Harvested from the household's crops: __________%

Other source (____________________): ___________%

***Total:* 100 %**

☐ 1 Don't know

Q.106 In the past 7 days, what foods did your household consume?

	CONSUMED IN LAST 7 DAYS	SOURCE OF FOOD			
	Yes	Free/donated	Purchased	Harvested	Don't know
Maize (posho)	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Matooke	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Sweet potatoes	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Irish potatoes	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Bread	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Rice	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Soybean	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Cassava	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Sorghum	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Groundnut	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Cowpea	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Beans	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Peas	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Yam	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Millet	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Green peppers	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Onions	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Tomatoes	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Eggplant	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Carrots	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Cabbage	☐ 1	☐ 1	☐ 2	☐ 3	☐ \4
Dodo	☐ 1	☐ 1	☐ 2	☐ 3	☐ \4
Okra	☐ 1	☐ 1	☐ 2	☐ 3	☐ \4
Sweet bananas	☐ 1	☐ 1	☐ 2	☐ 3	☐ \4
Mangoes	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Oranges	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Pineapple	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Passion fruit	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Papaya (pawpaw)	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4\
Beef	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Goat meat	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Pork	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Chicken	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Fish	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Eggs	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Milk	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Cooking oil	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Coffee	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4

Tea	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Soda	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4
Other: __________	☐ 1	☐ 1	☐ 2	☐ 3	☐ 4

Q.107 Before you moved here, what foods did your household typically consume at least once a week?

	TYPICALLY CONSUMED IN A WEEK	**SOURCE OF FOOD**		
	Yes	Free/donated	Purchased	Harvested
Maize (posho)	☐ 1	☐ 1	☐ 2	☐ 3
Matooke	☐ 1	☐ 1	☐ 2	☐ 3
Sweet potatoes	☐ 1	☐ 1	☐ 2	☐ 3
Irish potatoes	☐ 1	☐ 1	☐ 2	☐ 3
Bread	☐ 1	☐ 1	☐ 2	☐ 3
Rice	☐ 1	☐ 1	☐ 2	☐ 3
Soybean	☐ 1	☐ 1	☐ 2	☐ 3
Cassava	☐ 1	☐ 1	☐ 2	☐ 3
Sorghum	☐ 1	☐ 1	☐ 2	☐ 3
Groundnut	☐ 1	☐ 1	☐ 2	☐ 3
Cowpea	☐ 1	☐ 1	☐ 2	☐ 3
Beans	☐ 1	☐ 1	☐ 2	☐ 3
Peas	☐ 1	☐ 1	☐ 2	☐ 3
Yam	☐ 1	☐ 1	☐ 2	☐ 3
Millet	☐ 1	☐ 1	☐ 2	☐ 3
Green peppers	☐ 1	☐ 1	☐ 2	☐ 3
Onions	☐ 1	☐ 1	☐ 2	☐ 3
Tomatoes	☐ 1	☐ 1	☐ 2	☐ 3
Eggplant	☐ 1	☐ 1	☐ 2	☐ 3
Carrots	☐ 1	☐ 1	☐ 2	☐ 3
Cabbage	☐ 1	☐ 1	☐ 2	☐ 3
Dodo	☐ 1	☐ 1	☐ 2	☐ 3
Okra	☐ 1	☐ 1	☐ 2	☐ 3
Sweet bananas	☐ 1	☐ 1	☐ 2	☐ 3
Mangoes	☐ 1	☐ 1	☐ 2	☐ 3
Oranges	☐ 1	☐ 1	☐ 2	☐ 3
Pineapple	☐ 1	☐ 1	☐ 2	☐ 3
Passion fruit	☐ 1	☐ 1	☐ 2	☐ 3
Papaya (pawpaw)	☐ 1	☐ 1	☐ 2	☐ 3
Beef	☐ 1	☐ 1	☐ 2	☐ 3
Goat meat	☐ 1	☐ 1	☐ 2	☐ 3
Pork	☐ 1	☐ 1	☐ 2	☐ 3
Chicken	☐ 1	☐ 1	☐ 2	☐ 3
Fish	☐ 1	☐ 1	☐ 2	☐ 3
Eggs	☐ 1	☐ 1	☐ 2	☐ 3
Milk	☐ 1	☐ 1	☐ 2	☐ 3
Cooking oil	☐ 1	☐ 1	☐ 2	☐ 3
Coffee	☐ 1	☐ 1	☐ 2	☐ 3
Tea	☐ 1	☐ 1	☐ 2	☐ 3
Soda	☐ 1	☐ 1	☐ 2	☐ 3

Other: _________	□ 1	□ 1	□ 2	□ 3

Q.108 Before consuming food that is eaten uncooked (e.g., cabbage, mangos, oranges), which of the following is done to the food item?

□ 1 Rinsed with water
□ 2 Nothing (eaten as is)
□ 3 Other: _______________
□ 4 Don't know

Photographs E.1. Photographs of Bwaise, the Lubigi Sewage Treatment Plant, in-country university and non-governmental organization partners, and stakeholder discussions.


(a) Bwaise dwellings and a local drainage channel


(b) Pit latrine in Bwaise

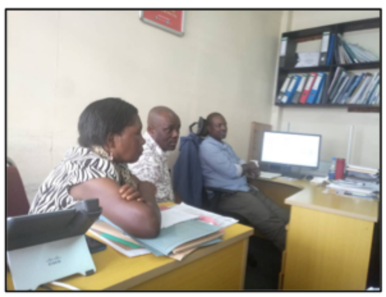

(c) Tanker truck discharging latrine sludge at Lubigi plant


(d) Collection of dried solids at Lubigi plant


(e) Survey enumerators from Makerere University with members of the UIUC research team


(f) Staff from Community Integrated Development Initiatives with members of the UIUC research team


(g) Discussions with Bwaise community leaders


(h) Discussions with staff from Kampala Capital City Authority


(i) Discussions with truck operators, facilitated by CIDI

Theme C. Quantitative modeling around sanitation alternatives in Bwaise, Uganda

Section E.1. General approach to quantitative scenario modeling. Details regarding the methodology and assumptions used to quantitatively model the resource recovery potential, net life cycle costs, and net greenhouse gas (GHG) emissions associated with each of the three sanitation system alternatives for Bwaise, Uganda are provided in the following sections. A number of tables (Tables E.6- E.11) present values, uncertainty ranges and distributions, and literature references for all parameters used in these analyses. To account for uncertainty and assess sensitivity of outcomes (e.g., costs, emissions, recovery potentials) to various input parameters, we employed a Monte Carlo analysis with Latin Hypercube Sampling (10,000 simulations). For each uncertain input parameter (Tables E.6-E.11; we calculated Spearman's rank correlation coefficients[184] to assess the sensitivity of each modeled outcome to that parameter.

Generally, models were implemented within a modular structure that separated each sanitation system alternative into distinct process stages related to the user interface, onsite storage, conveyance, centralized treatment and resource recovery, and reuse or disposal[84,264]. Each process stage could include separate modules for liquids and solids (if source separation is practiced onsite, or if separation processes occur during centralized treatment) or modules for combined excreta. Additionally, process stages focused on centralized treatment/resource recovery and reuse/disposal accommodated the possibility of multiple modules representing multi-stage treatment approaches (e.g., the existing system includes sedimentation followed by lagoons for liquid treatment and drying beds for solids management).

Section E.2. Model inputs and excretion estimates. Initial inputs common to all alternative systems included parameters regarding dietary intake, bodily waste excretion, and general assumptions related to biological degradation, the economic environment, and GHG emissions potential (Table E.6). The resource content of urine and feces (i.e., excretion rates of nitrogen, phosphorus, potassium, and chemical oxygen demand [COD]) was estimated from dietary intake following procedures used in previous work[29,167]. To summarize, per capita caloric and protein supply data for Uganda in 2013 were extracted from the UN Food and Agricultural Organization's statistical database (FAOSTAT)[53]. These data account for supply chain losses and wastes up to the household level. As household food waste is typically low in sub-Saharan Africa[323], we assumed household losses were negligible. To account for variations in diet across households and individuals, we defined an uncertainty range by increasing and decreasing the reported values by 10%. From protein and caloric intake, we then estimated nutrient intake. Nitrogen was computed as a fraction of total protein supply (13-19%)[57,58], while phosphorus was calculated using two separate fractions, depending on whether the protein is plant- or animal-based[29,58]. We estimated potassium using a factor converting from caloric intake to potassium[54,55]. Next, a fraction of intake is excreted. We assumed nitrogen and phosphorus excretion is at or near 100% of intake[52,324]. A lower excretion factor was used for potassium, because some ingested potassium (2-35%) leaves the body in sweat[54,55]. Finally, to estimate COD excretion, an excretion factor (2-10%)[17,20,50] was applied to caloric intake to estimate per capita energy excretion, and this energy value was converted to COD by calculating a lower heating value of wastewater (14 kJ·g COD^{-1}, assuming the organic matter in wastewater contains 50% protein, 40% carbohydrates, and 10% fats[5]). This process of estimating per capita resource excretion is represented by the equations below (Eqs. 1-4):

$$N_{cap} = (p_{tot})(N_{prot})(N_{exc}) \tag{1}$$

$$P_{cap} = \left[\left(P_{prot,v}\right)p_{veg} + \left(P_{prot,a}\right)p_{anim}\right]\left(P_{exc}\right) \quad (2)$$

$$K_{cap} = \left(e_{cal}\right)\left(K_{cal}\right)\left(K_{exc}\right) \quad (3)$$

$$COD_{cap} = \left(e_{cal}\right)\left(e_{exc}\right)\left(LHV_{WW}\right)^{-1} \quad (4)$$

where N_{cap}, P_{cap}, K_{cap}, and COD_{cap} represent per capita excretion of nitrogen, phosphorus, potassium, and COD (kg N·cap^{-1}·yr^{-1}, kg P·cap^{-1}·yr^{-1}, kg K·cap^{-1}·yr^{-1}, kg COD·cap^{-1}·yr^{-1}); p_{tot} represents total protein supply (kg·cap^{-1}·yr^{-1}); p_{veg} and p_{anim} represent vegetable and animal protein supplies, respectively (kg·cap^{-1}·yr^{-1}); e_{cal} represents caloric supply (kcal·cap^{-1}·yr^{-1}); N_{prot} represents the fraction of nitrogen contained in total protein; $P_{prot,v}$ and $P_{prot,a}$ represent the fractions of phosphorus contained in vegetable and animal protein, respectively; K_{cal} represents potassium content relative to caloric intake (kg K·kcal^{-1}); N_{exc}, P_{exc}, K_{exc}, and e_{exc} represent nitrogen, phosphorus, potassium, and energy excretion factors (total excreted in urine and feces as a fraction of intake); and LHV_{WW} represents wastewater's lower heating value (kJ·g COD^{-1}, which can be converted to kcal·kg COD^{-1} by multiplying by a unit conversion factor of 239).

These equations estimate total resource excretion in urine and feces. We used relevant literature[20,48] to identify typical fractions of each resource excreted in each waste stream (e.g., the percentage of total excreted nitrogen found in urine). From the literature, we also defined total excretion rates of urine and feces, the typical moisture content of each stream, and typical calcium and magnesium excretion rates in each stream[20,48]. The fraction of excreted nitrogen present in a reduced and inorganic form (urea or ammonia)[20,48,231] was also important to consider, as ammonia volatilization may cause substantial nitrogen losses during various processes.

Additional parameters related to economics (local currency exchange rate, discount/interest rate[96]), GHG emissions (e.g., equivalent 100-year CO_2 emissions associated with methane and nitrous oxide)[325], and the rate of biological degradation during storage were also included in this set of initial inputs. We assumed that degradation and transformation processes (e.g., breakdown of easily biodegradable COD, nitrification and denitrification) followed first-order reaction kinetics ($C = C_o e^{-kt}$), with "full" degradation (represented by 2-4 log reduction, as first-order exponential decay does not allow for complete 100% degradation) occurring after 1-3 years. From these assumptions, a first-order degradation rate constant can be calculated.

Section E.3. Bwaise's existing sanitation system. The existing sanitation system in Bwaise, Uganda includes pit latrines (typically shared by multiple households)[301,302], conveyance of latrine contents via tanker trucks, and centralized treatment at the Lubigi Sewage Treatment Plant (which involves sedimentation, lagoons for liquid treatment, and drying beds for solids management). Within the first process stage, the user interface combines all materials entering the latrine pit, including urine, feces, anal cleansing material (assumed to be toilet paper)[326,327], and flushing or cleaning water[235,328]. While toilet paper or other added materials may contribute non-negligible quantities of resources such as COD to the pit contents, we only tracked nutrients and COD contributed by urine and feces, to prevent our results from becoming dependent upon cleansing material or flush water composition. However, we did track the mass of each added material, which may increase transport requirements. Additionally, several parameters related to the materials, costs, and GHG emissions associated with latrine construction and operation were considered here (Table E.7). Construction and operating costs were based on previous analyses performed in the same context[315]. For this and subsequent process stages, an inventory of items generating GHG emissions was acquired using the ecoinvent v3.2 database[329] accessed within

SimaPro v8.5.2.0, a software for implementing life cycle assessment. Emissions were converted to equivalent kilograms of CO_2 using the U.S. EPA's Tool for the Reduction and Assessment of Chemicals and Other Environmental Impacts (TRACI; 2.1 v1.03, implemented within SimaPro v8.5.2.0)[330]. Within this process stage, we also estimated the number of users per latrine, based on our survey results related to household size (median of 4 people per household, with a standard deviation of 1.8) and latrine sharing (3-5 households per toilet). All model outcomes (resource recovery potential, costs, GHG emissions) for each scenario were normalized to report annual per capita results, using the number of users and an assumed latrine lifetime (5-10 years)[315,331]. Construction costs were annualized using the facility lifetime and an assumed annual discount rate (3-6%)[96].

In the onsite storage process stage, mixed excreta accumulate in the latrine pit and remain there until emptying. A great deal of uncertainty surrounds the processes occurring in this stage (e.g., water and nutrient infiltration into surrounding soil, ammonia volatilization, methane and CO_2 emissions from anaerobic and/or aerobic COD degradation, biological nitrogen transformation and N_2O emission from incomplete denitrification). We reviewed relevant literature to characterize the accumulation, losses, and emissions resulting from many of these processes, although reported estimates are often highly variable (Table E.8; e.g., nitrogen losses of 1-50% through leaching; sludge accumulation rates of 100-900 $L \cdot cap^{-1} \cdot yr^{-1}$)[231,301,307–309,328,332]. We also assumed 60-80% of COD was easily biodegradable (and would fully degrade due to biological activity if given sufficient time), while 70-90% of nitrogen was available for nitrification and denitrification processes[231]. For simplicity, we assumed all latrine pits were unlined and above the groundwater table, with 10-50% of COD degradation occurring anaerobically and 0.5-1.0% of transformed nitrogen being emitted as nitrous oxide[333]. To account for the non-steady state nature of pit latrine operation, we made several additional simplifying assumptions. First, we assumed that each new excreta deposit forms a layer that does not mix with previous deposits, and we assumed infiltration and volatilization processes occurred relatively quickly, happening before degradation and transformation processes (assumed to occur less rapidly). In each layer, easily biodegradable COD and available nitrogen degrade according to first-order reaction kinetics (assumptions used to estimate the rate constant are described in Section S2). When the collected waste is evacuated from storage, all layers are removed together, with each layer at a different point along the first-order decay curve. We assumed the COD and nitrogen concentrations in the evacuated waste can be represented as the average of the concentrations across all layers, calculated using the mean value theorem for integrals (i.e., integrating the first-order decay function from the start time to the emptying time, and dividing by the total elapsed time; Eq. 5).

$$C_{avg} = \frac{C_o}{k(t_f - t_o)}\left(e^{-kt_o} - e^{-kt_f}\right) \tag{5}$$

where C_{avg} represents the average concentration when the latrine is emptied; C_o represents the initial concentration; k is the first-order rate constant; and t_o and t_f represent the starting time and ending time, respectively. The differences between the initial concentrations and the averages calculated after storage were then used with the IPCC methodology[333] to calculate methane and N_2O emissions during the storage period. According to the IPCC methodology for methane, a given quantity of degraded COD is associated with a maximum potential methane production (0.175-0.325 kg $CH_4 \cdot kg\ COD^{-1}$). Based on environmental conditions, an appropriate methane correction factor (MCF) is then applied to estimate actual methane emissions. For example, a MCF of 0.25 signifies conditions where 25% of COD degradation occurs anaerobically. For N_2O, the emission factor (0.5-1.0%) is multiplied by total transformed nitrogen to estimate how much nitrogen leaves the system as N_2O.

The conveyance process stage involves pumping out the contents of full pit latrines into tanker trucks, which then transport the collected sludge to centralized treatment. Of the quantities of nutrients and COD remaining in the sludge after onsite storage, we assumed small fractions (≤5%) would be lost during pumping and transport. Based on our discussions with local stakeholders, we assume a charge of 20,000-44,000 Ugandan shillings (US $5-12) per cubic meter of sludge removed, with this fee covering all costs incurred by truck drivers (e.g., fuel, maintenance, personal protective equipment, discharge fees at the treatment plant, taxes). We assumed a transport distance of 2-10 kilometers (based on a Google Maps distance of 4-5 kilometers from Bwaise to the Lubigi Sewage Treatment Plant, with added uncertainty), and the emissions factor associated with transport (kilograms of equivalent CO_2 per tonne·kilometer) was based on factors reported for a variety of truck types and sizes[329,330]. The annual mass to be transported reflected the sludge accumulation rate during onsite storage, and the emptying period (i.e., the time between emptying events) was assumed to be roughly equivalent to latrine filling times (calculated from sludge accumulation rates and pit volumes estimated from survey results and relevant literature[301,302]).

The centralized treatment process stage is modeled after the Lubigi Sludge Treatment Plant, which includes sedimentation basins to separate solids and liquids, followed by unplanted drying beds for solids management and a series of lagoons (anaerobic and facultative) for liquid treatment. For the most part, the physical design (e.g., volumes, dimensions) of these treatment components was established based on discussions with Lubigi operators, process flow diagrams they provided, and direct visual assessment. Any performance data that could not be provided by the Lubigi operators was derived from relevant literature (Table E.10). Latrine sludge discharged from trucks enters one of two 1,250-m^3 sedimentation basins. These basins are used in an alternating fashion, with operation switching every 1-2 months to allow for removal of accumulated solids. We assumed a final solids content of settled sludge of 10-20%, with total solids retention of 50-80% and COD retention of 70-95%[306,328]. We assumed the fractions of nutrients retained in settled sludge were roughly equivalent to the fractions of total excreted nutrients present in feces[48]. Any COD degradation occurring in the settled sludge before it is removed from the basin was assumed to be predominantly (80-100%) anaerobic, and N_2O emissions were expected to be relatively low (0.05-0.6% of available nitrogen), as nitrification requires oxygen as the electron acceptor[333]. After removal from the sedimentation basin, settled sludge is stored in covered and uncovered drying beds for a period of at least 6 months, where sludge may reach a final solids content of up to 90%[334,335]. Lubigi houses a total of 19 sludge storage beds (deeper than drying beds to provide additional storage capacity), 19 covered drying beds, and 30 uncovered drying beds. We assumed drying beds provide a relatively aerobic environment (0-30% of COD degrading anaerobically), which also suggests that nitrification/denitrification processes may be more substantial here (0.5-1.0% available nitrogen emitted as N_2O)[333]. Liquids leaving the sedimentation basin enter three 4,640-m^3 anaerobic lagoons operated in parallel, which may remove 50-85% of influent COD[294,306]. We assumed this removed COD settles, and the easily biodegradable fraction degrades over time in the lagoon's anaerobic environment. Following the anaerobic lagoons are two 11,530-m^3 facultative lagoons, also operated in parallel. COD removal efficiencies of 80-95% are expected[306], with removed COD degrading over time in a predominantly aerobic environment. Overall, Lubigi operators report that the total installation cost of the plant was approximately USD 18.6 million. It is expected to operate for 8-11 years and currently serves 30,000 sewered customers (below its planned design population of 50,000). Additionally, the plant manages approximately 500 $m^3 \cdot d^{-1}$ of sludge. Only sludge enters the sedimentation basin, while the lagoons treat wastewater as well as the liquid effluent from sedimentation. Based on CIDI's expectations regarding the alternative plant's daily sludge flow rate and population served, we estimated the total population of latrine users served by the existing plant. To estimate GHG emissions from construction, we approximated the quantities of key materials (e.g., concrete, metal roofing, structural steel, plastic lagoon liners, excavation) that would have been necessary to build these treatment processes and obtained relevant emissions impact factors[329,330]. Limitations on data availability prevented a complete life cycle assessment that incorporated all construction materials, but our results (using the key materials listed above, which likely represent the majority of construction emissions) showed that GHG emissions from treatment plant construction are minor when compared with other emissions categories (e.g., direct emissions from excreta degradation, latrine construction). Lubigi operators report the plant requires 4,760 kWh of electricity per month, and we assume that electricity is the main contributor to operational costs and GHG emissions. We applied a unit electricity cost based on various categories of local tariffs (USD 0.08-0.21$\cdot$kWh^{-1}) and a unit GHG impact factor of 0.106-0.121 kg CO_2eq$\cdot$kWh^{-1} to estimate emissions from electricity. This GHG factor is based on Uganda's electricity mix, which is dominated by hydroelectric power[329,336]. The plant's operating expenses also include salaries for twelve employees.

In the reuse or disposal process stage, we assumed that farmers purchase all dried solids, and that all treated liquid effluent is used to irrigate cropland. In reality, the plant discharges liquid effluent into local wetlands, but some crop production occurs in this ecosystem. To estimate the maximum benefits from resource recovery in this existing scenario, we assume both liquids and solids are purchased by farmers for cropland application. The transfer process (e.g., farmers load dried solids onto trucks for transport to farms) is assumed to result in some small resource losses (Table E.11)[61], and we assume the purchase price depends upon nutrient content. We collected prices of several single-nutrient fertilizers (e.g., urea, triple superphosphate, potassium chloride) in Kampala and also used available data on national retail prices in Uganda from 2010 to 2017 to estimate a price per kilogram of each nutrient (nitrogen, phosphorus, and potassium)[337]. We then used our median estimates of the nutrient content in dried solids to compute the price per tonne of solids, if the nutrients in dried solids are given the same value as the nutrients in retail fertilizers. Finally, we compared this price estimate with the range of solids selling prices currently reported by the Lubigi plant (18,000-60,000 Ugandan shillings per tonne of dried sludge, or approximately US \$5-17$\cdot$tonne^{-1}). Through this comparison, we calculated a discount factor, representing the inconveniences and reductions in perceived value associated with using dried sludge rather than retail fertilizers (e.g., dried sludge is bulkier, requires more energy for transport, and may contain more variable quantities of nutrients). By multiplying the estimated nutrient concentrations in dried sludge by their fertilizer unit prices (per kilogram of nutrient) and this discount factor, we estimated the value of sludge produced at the treatment plant. Due to a lack of data on the value of organic matter independent of nutrients, we note that this procedure does not account for the organic content of the sludge, which may also offer agricultural benefits when land applied. Also due to a lack of relevant data, we assumed that nutrients recovered in liquid effluent can be valued similarly to those in dried sludge, with the same discount factor. This calculated value from recovered solids and liquids represents an economic benefit from resource recovery and reuse, and it functions to reduce the overall net costs associated with this sanitation alternative. Analogously, land application of these resources can offset GHG emissions that would have resulted from the production of fertilizers (if we assume sludge application replaces existing or potential future fertilizer use). We identified emissions factors (equivalent CO_2 emissions per kilogram of nitrogen, phosphorus, or potassium produced) associated with the production of several single-nutrient fertilizers (e.g., urea, ammonium nitrate, single superphosphate, triple superphosphate, potassium chloride)[329,330], using these to estimate offsets associated with the nutrients embedded in dried sludge and liquid effluent. These offsets functioned to lessen the overall net GHG emissions of the sanitation system.

Section E.4. CIDI's proposed treatment center. The second sanitation alternative employs the same onsite latrines and truck conveyance processes as the existing system. Therefore, the assumptions and procedures relevant for the user interface, onsite storage, and conveyance process stages do not change. The difference comes in the centralized treatment processes that occur. The proposed treatment center includes anaerobic digestion of mixed waste in a three-chambered anaerobic baffled reactor (including a filter at the end), followed by unplanted and planted drying beds for solids management and a planted bed for secondary treatment of liquids. We designed all components using relevant literature and information from CIDI (Table E.10). The anaerobic baffled reactor (with filter) was designed with an overall hydraulic retention time of 1-5 days[5,306] and is expected to remove 83-99% of influent COD and 0-15% of total nitrogen[5,306]. As with the existing treatment system, we approximated the quantities of key materials needed to build these alternative treatment facilities (e.g., concrete, steel, filter media), combining these quantities with impact factors to estimate GHG emissions associated with construction. Regarding operation, we estimated electricity use based on Lubigi's consumption, scaled according to the plant's daily sludge capacity (as CIDI does not yet have precise estimates of operational needs). Construction costs were based on estimates performed by CIDI engineers, and operating expenses (including labor and electricity) were calculated based on approximated electricity consumption and staffing expectations. The overall system has a treatment capacity of 60 m^3 of sludge per day, which is considerably lower than the capacity of the existing Lubigi plant. The expected lifetime of the system is reported to be approximately 50 years, substantially longer than the existing plant's expected lifetime. Normalizing all outcomes on an annual, per capita basis allows for direct comparison between these two alternatives, although it is important to note their differences in scale.

Assumptions around the reuse of recovered materials were similar to those for the existing system, with cost and emissions offsets depending upon the nutrient content of products. With regard to methane-rich biogas recovered from anaerobic digestion, losses during operation, collection, and storage may range from negligible levels in well-designed and efficiently managed systems to 20% under less ideal circumstances[151]. We assume the value will fall somewhere within this 0-20% range. As liquid petroleum gas (LPG) is likely the closest analog to biogas readily available in this context, we assume the energetic content of recovered methane will have a similar economic value to an equivalent quantity of energy provided in LPG tanks currently available for purchase. Based on price data collected from local vendors, the cost to refill an empty LPG tank (after the tank had already been purchased) is estimated to be 6,000-6,700 Ugandan shillings per kilogram (US \$1.50-1.90$\cdot$kg LPG^{-1}), while the specific energy of LPG[338] is reported to be approximately 50 $MJ \cdot kg^{-1}$. This income from the sale of biogas represents a reduction in the system's total net costs. Similarly, we assume that combustion of biogas for cooking will offset LPG combustion. While 90% of surveyed households in Bwaise reported using charcoal as their cooking fuel, we focus on a comparison between biogas and LPG because these alternatives likely provide similar levels of service and indoor air quality (both biogas and LPG are reported to emit less particulate matter than charcoal)[294,339]. Furthermore, comparing biogas with LPG (as opposed to charcoal) represents a more conservative approach to estimating GHG offsets, as LPG contributes smaller quantities of GHG emissions than charcoal not originating from sustainably managed forests (deforestation rates and diminishing areas of forested land in Uganda suggest forests are often not managed sustainably), especially when charcoal is burned in inefficient traditional stoves[53,340]. We assume that LPG cooking stoves have roughly the same efficiency as biogas stoves[151,340], and that approximately three kilograms of carbon dioxide equivalents are emitted from burning one kilogram of LPG[341]. We calculate offsets converting the energy recovered in biogas to an equivalent mass of LPG and multiplying by the emissions factor of 3 kg $CO_2eq \cdot kg\ LPG^{-1}$.

Section E.5. Container-based sanitation. The final system alternative replaces existing pit latrines with a container-based approach. Toilet facilities associated with container-based sanitation typically separate urine and feces, storing each material in a removable container that can be collected and replaced with an empty one. We assume the costs and greenhouse gas emissions associated with the construction of a container-based toilet facility will be similar to those of a urine-diverting dry toilet[315]. We assume users add a desiccant (e.g., wood ash) to the fecal collection container after each toilet use (200-500 $mL \cdot cap^{-1} \cdot d^{-1}$) to reduce moisture content and control odors[294], and moisture content decreases over time down to an absolute minimum of 7-13% according to an estimated first-order exponential decay constant (0.009-0.011 d^{-1})[342]. The feces container is assumed to be mostly aerobic (0-20% of COD degradation is anaerobic), such that potential N_2O emissions (0.5-1.0% of available nitrogen) are somewhat higher than in pit latrines.

When urine is stored, the pH increase driven by the hydrolysis of urea induces precipitation of minerals (primarily struvite and hydroxyapatite when collected urine is not diluted)[311,313]. Struvite ($MgNH_4PO_4 \cdot 6H_2O$) forms from magnesium, ammonium, and phosphate ions, and is typically limited by magnesium concentrations in undiluted urine. Precipitation of hydroxyapatite ($Ca_5(PO_4)_3OH$) tends to be limited by calcium concentrations[311–313]. These processes occur relatively quickly (i.e., within less than a few days)[311,312], so we assume they have already reached equilibrium when containers are collected for transport to the centralized facility. Based on influent concentrations of constituent ions, we estimate the quantity of struvite precipitated at equilibrium using a conditional solubility product (pK_{sp}^{cond} = 7.3-8.1) estimated for typical applications involving source-separated urine (pH = 9, temperature = 25°C, ionic strength = 0.16-0.61)[287]. As hydroxyapatite's solubility product is reported to be quite small (pK_{sp} = 57.5)[311], we assumed that calcium ions in solution (which are typically the limiting constituent in undiluted urine) would be negligible after equilibrium has been reached. In other words, hydroxyapatite precipitation would incorporate all calcium ions and the corresponding quantity of phosphate ions, based on the mineral's molar ratio. In urine storage containers, most precipitated minerals are reported to settle and contribute to the formation of a viscous sludge, which can be recovered when urine is removed[61,313]. However, we do assume that an uncertain fraction (0-50%) of precipitated minerals forms a hard scale on the walls of the storage container[313], which may be more difficult to remove. We treat this fraction as a loss that is unavailable for recovery. We also assume that 0-7% of total nitrogen is lost through ammonia volatilization[61,231,343].

In many cases, container-based systems employ one or more manual pushcarts to collect urine and feces containers from each toilet facility and provide users with clean, empty containers. An analysis that modeled this type of collection system in similar contexts (informal settlements in Kampala, Uganda and Raipur, India) estimated costs to be US \$0.004-0.015$\cdot cap^{-1} \cdot d^{-1}$ when the pushcart system operated under a performance-based payment scheme (i.e., workers are paid according to the number of serviced facilities) and served at least 200 users (costs under a fixed payment scheme, where workers receive a fixed daily payment, were similar when the system served at least 700 individuals)[96]. We assume pushcart collection costs will fall within a similar range in Bwaise. However, we also assume an additional transport step to convey collected containers to a centralized treatment facility such as the Lubigi plant. Pushcarts load collected containers onto a truck parked in the settlement, and then this truck moves the containers to the treatment plant. As with the other sanitation alternatives, we assume that small losses of nutrients and COD (≤5%) may occur during these conveyance processes, although losses may be less likely in this case, since the waste is confined within closed containers. Based on CIDI's local experience, trucking costs were estimated to be \$3-11$\cdot m^{-3}$ of collected waste. GHG emissions associated with transporting a given mass by truck were assumed to be similar to those in the preceding alternatives.

After conveyance, we assumed that centralized treatment and recovery processes are the same as those employed in the existing system alternative, enabling us to investigate the specific impact of replacing pit latrines with container-based facilities. Within the existing treatment approach, sedimentation is no longer necessary for separation of solids and liquids, because feces and urine already arrive at the treatment plant in separate containers. Accordingly, sedimentation is removed from the treatment sequence. We acknowledge that the existing treatment system (drying beds, wastewater lagoons) may not be the most appropriate approach for the desiccated feces and stored urine that will now be entering the plant. However, we continue to employ these processes because we have applicable cost data from the existing plant and more appropriate alternatives (e.g., composting of solids, extended storage of urine) may be associated with similar infrastructure. While our goal with this scenario was to understand the specific implications of transitioning from pit latrines to container-based facilities within the existing sanitation system, future work should examine the downstream process changes that may be needed to most effectively accommodate the different inputs entering centralized treatment. Reuse assumptions surrounding crop application of dried solids and treated liquid effluent were also the same as in the existing system.

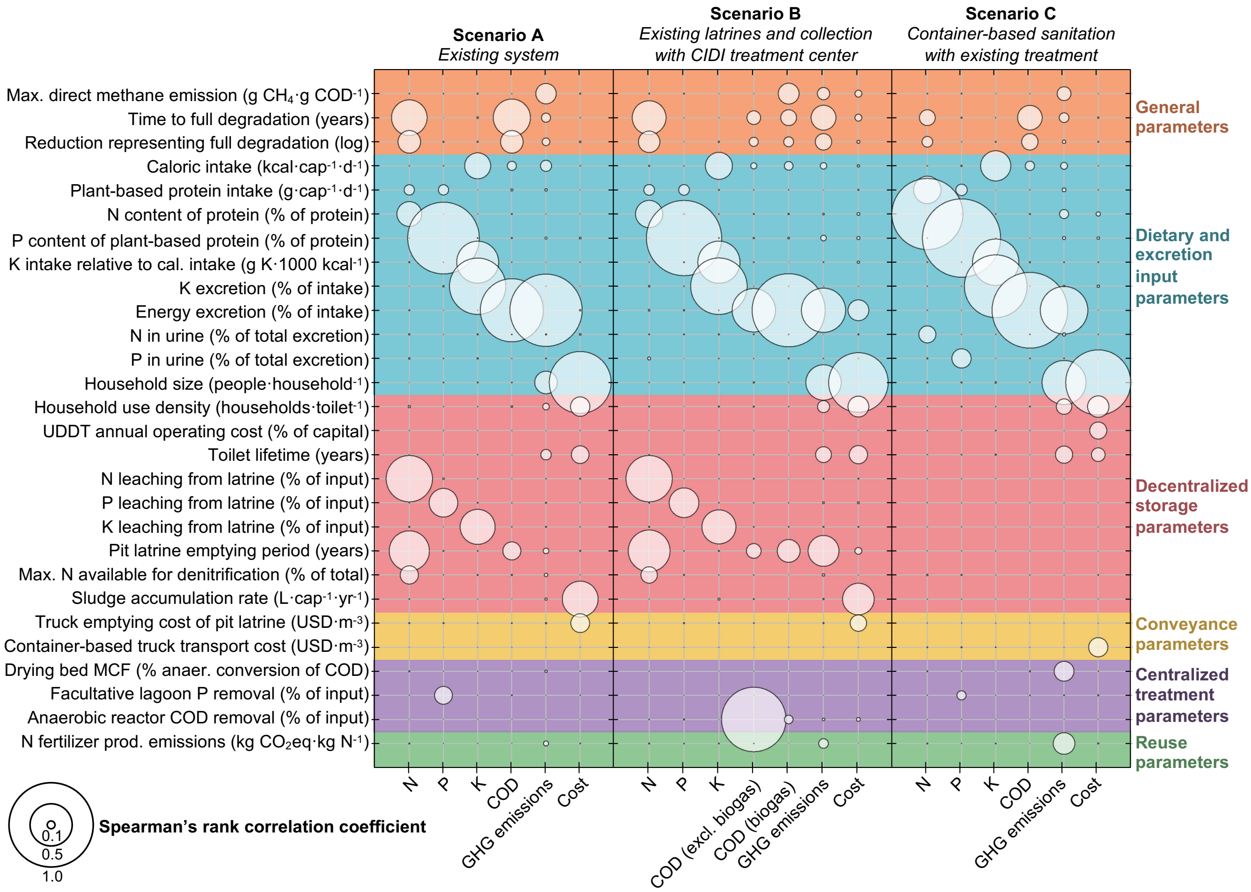


Figure E.1. The relative importance of key parameters to the uncertainty associated with the outputs of each scenario (N, P, K, and COD recovery potentials; net GHG emissions; net costs). The size of each bubble is proportional to the absolute value of the Spearman's rank correlation coefficient associated with each parameter and output combination, calculated using the results from 10,000 simulations included in the uncertainty analysis. Spearman's coefficients estimate the degree to which variations in an output have a monotonic relationship with variations in an input parameter's value. Each parameter shown in this figure has a coefficient with an absolute value of at least 0.20 for at least one output in at least one scenario. Parameters are divided into categories according to their stage along the sanitation chain (background shading).

Table E.5. Summary of alternative sanitation scenarios modeled for Bwaise, Uganda. Each sanitation alternative is described as an interconnected sequence of processes. Cells with gray shading and white text indicate differences from the existing sanitation system.

Scenario	User interface	Onsite storage/treatment	Conveyance	Centralized treatment/recovery	Reuse or disposal
(a) Existing system	Dry or pour-flush toilet	Single latrine pit	Tanker truck for pumping out and transporting latrine contents when pit is full	Sedimentation; covered and uncovered drying beds (solids); anaerobic and facultative lagoons (liquids)	Cropland application of treated solids and liquids
(b) CIDI treatment center	Dry or pour-flush toilet	Single latrine pit	Tanker truck for pumping out and transporting latrine contents when pit is full	Anaerobic baffled reactor; unplanted and planted drying beds (solids); planted bed (liquids)	Cropland application of treated solids and liquids; biogas collection for cooking fuel
(c) Container-based sanitation	Urine-diverting dry toilet	Liquid container; solids container (with desiccant addition)	Container collection with manual push-carts, transferred to truck for transport to treatment facility	Covered and uncovered drying beds (solids); anaerobic and facultative lagoons (liquids)	Cropland application of treated solids and liquids

Table E.6. Parameter values, ranges, and distributions used as initial inputs common to all scenarios in the quantitative modeling analysis.

Parameter	Expected value	Low value	High value	Distribution	References
Caloric intake (kcal·cap^{-1}·d^{-1})	2,130	1,917	2,343	uniform	[53]
Vegetable protein intake (g·cap^{-1}·d^{-1})	40.29	36.26	44.32	uniform	[53]
Animal protein intake (g·cap^{-1}·d^{-1})	12.39	11.15	13.63	uniform	[53]
N content of protein (%)	13	13	19	uniform	[57,58]
P content of veg. protein (%)	2.2	0.4	4.8	triangular	[56,58]
P content of animal protein (%)	1.1	0.2	3.2	triangular	[56,58]
K content of cal. intake (g K·1000 kcal^{-1})	1.2	1.1	1.5	uniform	[54,55]
N excretion (% of intake)	100	99	100	uniform	[52,324]
P excretion (% of intake)	100	99	100	uniform	[52,324]
K excretion (% of intake)	88	65	98	uniform	[54,140]
Energy excretion (% of intake)	6	2	10	uniform	[17,20,50]
N in urine (% of total)	88	74	93	triangular	[20,48]
P in urine (% of total)	61	33	75	triangular	[20,48]
K in urine (% of total)	74	53	93	triangular	[20,48]
Energy in feces (% of total)	81	69	90	triangular	[20,48]
Urine N in reduced inorganic form (%)	85	75	90	uniform	[20,48]
Feces N in reduced inorganic form (%)	20	16	24	uniform	[20,231]
Urine excretion (g·cap^{-1}·d^{-1})	1,400	800	2,500	triangular	[20,48]
Feces excretion (g·cap^{-1}·d^{-1})	250	75	520	triangular	[20,48]
Urine moisture content (%)	95	93	97	triangular	[20,48]
Feces moisture content (%)	85	76	88	triangular	[20,48]
Mg in urine (g Mg·cap^{-1}·d^{-1})	0.2	0.12	0.21	uniform	[48,311,312]
Mg in feces (g Mg·cap^{-1}·d^{-1})	0.25	0.15	0.34	uniform	[48]
Ca in urine (g Ca·cap^{-1}·d^{-1})	0.28	0.057	0.5	uniform	[48]
Ca in feces (g Ca·cap^{-1}·d^{-1})	1.9	0.1	3.6	uniform	[48]
Max. methane emission (g CH_4·g COD^{-1})	0.25	0.175	0.325	triangular	[333]
Time to full degradation (years)	2	1	3	uniform	(assumption)
Reduction rep. full degradation (log units)	3	2	4	uniform	(assumption)
N_2O GWP (kg CO_2eq·kg N_2O^{-1})	265	265	298	uniform	[325]
CH_4 GWP (kg CO_2eq·kg CH_4^{-1})	28	28	34	uniform	[325]
Exchange rate (UGX·USD^{-1})	3,700	3,600	3,900	triangular	Bank of Uganda, 2019
Annual discount rate (%)	5	3	6	uniform	[96]

Table E.7. Parameter values, ranges, and distributions used in the user interface process stage of the quantitative modeling analysis. Parameters related to pit latrines are used in Scenarios A-B, while those related to urine-diverting dry toilets (UDDTs) are used in Scenario C (Table E.5).

Parameter	Expected	Low	High	Dist.	References
Toilet paper addition (sheets·cap^{-1}·d^{-1})	12.4	11.7	14.2	uniform	[326,327]
Toilet paper solid mass (mg·sheet^{-1})	545	511	578	uniform	[326,327]
Water addition (L·cap^{-1}·d^{-1})	10	4	25	triangular	[235,328]
UDDT desiccant volume (mL·cap^{-1}·d^{-1})	200	200	500	triangular	[294]
UDDT desiccant density (kg·m^{-3})	760	663	977	triangular	[344,345]
UDDT desiccant Mg content (% of total mass)	2.24	0.8	5.62	triangular	[344,346–350]
UDDT desiccant Ca content (% of total mass)	30.34	7.42	37.16	triangular	[344,346–350]
Household size (cap·household^{-1})	4 (1.8 st. dev.)	1	-	normal	(survey results)
Household use density (households·toilet^{-1})	4	3	5	uniform	(survey results)
Pit latrine capital cost (USD·toilet^{-1})	449	386	511	uniform	[315]
Pit latrine annual oper. cost (% of capital)	5	2	8	uniform	[331]
UDDT capital cost (USD·toilet^{-1})	553	476	630	uniform	[315]
UDDT annual operating cost (% of capital)	10	5	15	uniform	[285]
Toilet lifetime (years)	8	5	10	uniform	[315]
Pit latrine construction materials					
Cement (kg)	700	-	-	-	[315]
Sand (m^3)	2.2	-	-	-	[315]
Gravel (m^3)	0.8	-	-	-	[315]
Bricks	54	-	-	-	[315]
Plastic sheet (m^2)	16	-	-	-	[315]
Steel (m^3)	0.00425	-	-	-	[315]
Excavation (m^3)	3.66	-	-	-	[315]
Wood (m^3)	0.19	-	-	-	[315]
UDDT construction materials					
Cement (kg)	200	-	-	-	[315]
Sand (m^3)	0.6	-	-	-	[315]
Gravel (m^3)	0.2	-	-	-	[315]
Bricks	682	-	-	-	[315]
Plastic sheet (m^2)	4	-	-	-	[315]
Steel (m^3)	0.00351	-	-	-	[315]
Stainless steel sheet (m^2)	28.05	-	-	-	[315]
Wood (m^3)	0.222	-	-	-	[315]
Material properties					
Plastic sheet mass (kg·m^{-2})	0.63	0.31	1.24	uniform	Colorado Lining International
Brick volume (m3·brick^{-1})	0.0024	-	-	-	(assumption)
Brick density (kg·m^{-3})	1750	1500	2000	uniform	(assumption)
Steel sheet mass (kg·m^{-2})	2.64	2.26	3.58	uniform	Home Depot
Gravel bulk density (kg·m^{-3})	1600	1520	1680	uniform	(assumption)
Sand bulk density (kg·m^{-3})	1442	1281	1602	uniform	(assumption)
Steel density (kg·m^{-3})	7900	7750	8050	uniform	(assumption)
Greenhouse gas unit impact factors					
Steel (kg CO_2eq·kg^{-1})	2.55	2.13	3.15	uniform	[329]
Stainless steel (kg CO_2eq·kg^{-1})	4.33	3.07	5.5	uniform	[329]
Stainless steel sheet rolling (kg CO_2eq·kg^{-1})	0.65	0.58	0.71	uniform	[329]
Excavation (kg CO_2eq·m^{-3})	0.53	0.51	0.55	uniform	[329]
Plastic (kg CO_2eq·kg^{-1})	1.97	1.93	2.01	uniform	[329]
Gravel (kg CO_2eq·kg^{-1})	0.015	0.012	0.018	uniform	[329]
Sand (kg CO_2eq·kg^{-1})	0.012	0.011	0.013	uniform	[329]
Cement (kg CO_2eq·kg^{-1})	1.08	0.97	1.19	uniform	[329]
Bricks (kg CO_2eq·kg^{-1})	0.28	0.25	0.31	uniform	[329]
Wood (kg CO_2eq·m^{-3})	197	186	208	uniform	[329]

Table E.8. Parameter values, ranges, and distributions used in the decentralized storage process stage of the quantitative modeling analysis. Parameters related to latrine pits are used in Scenarios A-B, while those related to urine storage tanks and feces containers are used in Scenario C (Table E.5).

Parameters	Expected	Low	High	Distribution	References
Single latrine pit					
Pit volume (m^3)	3.66	-	-	-	(survey results; assumptions)
Pit emptying period (yr)	0.8	0.3	2.4	triangular	(survey results; model calculations)
Sludge accumulation rate ($L \cdot cap^{-1} \cdot yr^{-1}$)	270	100	900	triangular	[328,332]
N leaching (% of input)	13	1	50	uniform	[231,301,307–309]
P leaching (% of input)	18	0	37	uniform	[231,301,307]
K leaching (% of input)	21	11	31	uniform	[307]
N_2O emission factor (% of degraded N)	0.8	0.5	1.0	triangular	[333]
N volatilization (% of input)	0.5	0	1	uniform	[231,301,308]
Methane correction factor (% anaerobic conversion of degraded COD)	35	10	50	triangular	[333]
Urine storage tank					
N volatilization (% of total)	5	0	7	uniform	[231,343]
Struvite conditional pK_{sp}	7.57	7.3	8.1	uniform	[287]
Precipitate sludge (% of precipitate that settles and can be removed)	75	50	100	uniform	[61,313]
Feces container					
Minimum feces moisture content after extended storage (%)	10	7	13	uniform	[342]
Moisture content exponential decay rate constant (d^{-1})	0.01	0.009	0.011	uniform	[342]
Methane correction factor (% anaerobic conversion of degraded COD)	10	0	20	triangular	[333]
N_2O emission factor (% of degraded N)	0.9	0.5	1	triangular	[333]
General parameters					
Maximum COD removal (% of input)	70	60	80	triangular	[231]
Maximum N degradation (% of input)	80	70	90	triangular	[231]
Container-based collection period (d)	3.5	1	9	triangular	[96]

Table E.9. Parameter values, ranges, and distributions used in the conveyance process stage of the quantitative modeling analysis.

Parameter	Expected	Low	High	Distribution	References
Tanker truck parameters					
N loss (% of input)	2	0	5	uniform	(assumption)
P loss (% of input)	2	0	5	uniform	(assumption)
K loss (% of input)	2	0	5	uniform	(assumption)
Mg loss (% of input)	2	0	5	uniform	(assumption)
Ca loss (% of input)	2	0	5	uniform	(assumption)
C loss (% of input)	2	0	5	uniform	(assumption)
Transport distance (km)	5	2	10	uniform	(assumption)
Emission factor (kg $CO_2eq \cdot t^{-1} \cdot km^{-1}$)	0.194	0.0576	0.526	uniform	[329]
Latrine emptying cost ($UGX \cdot m^{-3}$)	32,000	20,000	44,000	uniform	CIDI; truck operators
Handcart and truck parameters					
N loss (% of input)	2	0	5	uniform	(assumption)
P loss (% of input)	2	0	5	uniform	(assumption)
K loss (% of input)	2	0	5	uniform	(assumption)
Mg loss (% of input)	2	0	5	uniform	(assumption)
Ca loss (% of input)	2	0	5	uniform	(assumption)
C loss (% of input)	2	0	5	uniform	(assumption)
Transport distance (km)	5	2	10	uniform	(assumption)
Emission factor (kg $CO_2eq \cdot t^{-1} \cdot km^{-1}$)	0.194	0.0576	0.526	uniform	[329]
Container collection cost ($USD \cdot cap^{-1} \cdot d^{-1}$)	0.010	0.004	0.015	uniform	[96]
Truck transport cost ($UGX \cdot m^{-3}$)	25,600	14000	39,600	uniform	(CIDI)

Table E.10. Parameter values, ranges, and distributions used in the centralized treatment process stage of the quantitative modeling analysis. Sedimentation, anaerobic and facultative lagoon, and unplanted drying bed parameters are used in the existing treatment plant (Scenarios A and C), while anaerobic digestion, anaerobic filter, and unplanted drying bed parameters are used in the alternative plant (Scenario B).

Parameter	expected	low	high	distribution	References
Sedimentation parameters					
Solids residence time (days)	45	30	60	uniform	(Lubigi)
Final solids content (%)	14	10	20	uniform	[306]
Solids retention (% of input)	50	35	60	uniform	(Lubigi)
COD retention (% of input)	50	35	60	uniform	(Lubigi)
Maximum COD degradation (% of retained)	70	60	80	triangular	[231]
Methane correction factor (% anaerobic conversion of degraded COD)	80	80	100	triangular	[333]
Maximum N degradation (% of retained N)	80	70	90	triangular	[231]
N_2O emission factor (% of degraded N)	0.5	0.05	0.6	triangular	[333]
N retention (% of input)	6	2.45	15.6	triangular	Calculations; [48]
P retention (% of input)	19.5	8.75	40.2	triangular	Calculations; [48]
K retention (% of input)	13	2.45	28.2	triangular	Calculations; [48]
Mg retention (% of input)	28	19	37	uniform	Calculations; [48]
Ca retention (% of input)	44	22	53	uniform	Calculations; [48]
Volume (m^3)	1,250	-	-	-	(Lubigi)
Length:width ratio	3.3	3.0	3.5	uniform	(Lubigi)
Average width:height ratio	3.6	3.3	3.8	uniform	(Lubigi)
Number of tanks	2	-	-	-	(Lubigi)
Columns per side	12.0	-	-	-	(Lubigi)
Anaerobic lagoon parameters					
COD removal (% of input)	70	60	70	triangular	Lubigi; [294]
Maximum COD degradation (% of retained)	70	60	80	triangular	[231]
Methane correction factor (% anaer. conv. of deg. COD)	80	80	100	triangular	[333]
Volume (m^3)	4,640	-	-	-	(Lubigi)
Length (m)	65	-	-	-	(Lubigi)
Width (m)	30	-	-	-	(Lubigi)
Number of lagoons	3	-	-	-	(Lubigi)
Facultative lagoon parameters					
COD removal (% of input)	70	70	90	triangular	Lubigi; [294]
Maximum COD degradation (% of retained)	70	60	80	triangular	[231]
Methane correction factor (% anaer. conv. of deg. COD)	20	0	30	triangular	[333]
Maximum N degradation (% of input)	80	70	90	triangular	[231]
N_2O emission factor (% of degraded N)	0.8	0.5	1	triangular	[333]
P removal (% of input)	60	50	70	uniform	(Lubigi)
Volume (m^3)	11,530	-	-	-	(Lubigi)
Length (m)	170	-	-	-	(Lubigi)
Width (m)	50	-	-	-	(Lubigi)
Number of lagoons	2	-	-	-	(Lubigi)
Unplanted drying bed parameters					
Retention time (days)	180	180	270	triangular	(Lubigi)
Final solids content (%)	35	30	40	uniform	(Lubigi)
Maximum COD degradation (% of total)	70	60	80	triangular	[231]
Methane correction factor (% anaer. conv. of deg. COD)	20	0	30	triangular	[333]
Maximum N degradation (% of input)	80	70	90	triangular	[231]
N_2O emission factor (% of degraded N)	0.8	0.5	1	triangular	[333]
Number of covered drying beds	19	-	-	-	(Lubigi)
Number of uncovered drying beds	30	-	-	-	(Lubigi)
Number of storage beds	19	-	-	-	(Lubigi)
Storage bed wall height (m)	1.5	1.2	1.8	uniform	(Lubigi)
Covered bed width (m)	7	-	-	-	(Lubigi)
Covered bed length (m)	34	-	-	-	(Lubigi)
Drying bed wall height (m)	0.6	0.45	0.75	uniform	(Lubigi)
Uncovered bed width (m)	7	-	-	-	(Lubigi)
Uncovered bed length (m)	31	-	-	-	(Lubigi)
Columns per side in covered beds	7	-	-	-	(Lubigi)
Column height in covered beds (m)	2.75	2.5	3	uniform	(Lubigi)
Steel column mass ($kg \cdot m^{-1}$)	30	23	37	uniform	(Sandeep Steels)

Table E.10 (cont.)

Parameter	expected	low	high	distribution	References
Anaerobic baffled reactor parameters					
Hydraulic retention time (days)	3	1	5	uniform	CIDI; [5,306]
COD removal (% degraded)	93	83	99	uniform	[5]
N removal (% of input N)	7	0	15	uniform	[306]
Length (m)	17	-	-	-	(CIDI)
Width (m)	5	-	-	-	(CIDI)
Height (m)	2.5	2.0	3.0	uniform	(CIDI)
Baffles	2	-	-	-	(CIDI)
Additional concrete for receiving basin, etc. (%)	25	20	30	Uniform	(assumption)
Liquid treatment bed parameters					
Hydraulic retention time (days)	3	1	5	uniform	CIDI; [5,306]
COD removal (% degraded)	70	50	90	uniform	[306]
Methane correction factor (% anaer. conv. of deg. COD)	80	80	100	triangular	[333]
Maximum N degradation (% of input)	80	70	90	triangular	[231]
N_2O emission factor (% of degraded N)	0.5	0.05	0.6	triangular	[333]
Length (m)	17.49	-	-	-	(CIDI)
Width (m)	12.415	-	-	-	(CIDI)
Height (m)	1.5	1.2	1.8	Uniform	(CIDI)
Alternate drying beds parameters (other parameters are the same as existing drying beds)					
Final solids content (%)	55	40	70	uniform	[306]
Length (m)	22.345	-	-	-	(CIDI)
Width (m)	19.33	-	-	-	(CIDI)
Height (m)	0.60	0.45	0.75	Uniform	(CIDI)
General parameters					
Methane energetic content ($kJ \cdot mol\ CH_4^{-1}$)	803	802	870	triangular	[5,16,351]
Sewer flow to existing plant ($m^3 \cdot d^{-1}$)	2,750	2,500	3,000	uniform	(Lubigi)
Latrine sludge flow to existing plant ($m^3 \cdot d^{-1}$)	500	-	-	-	(Lubigi)
Latrine sludge flow to alternative plant ($m^3 \cdot d^{-1}$)	60	-	-	-	(CIDI)
Concrete thickness (m)	0.3	0.15	0.45	uniform	(assumption)
Plastic liner mass ($kg \cdot m^{-2}$)	0.63	0.31	1.24	uniform	Colorado Lining International
Gravel bulk density ($kg \cdot m^{-3}$)	1600	1520	1680	uniform	
Roof slope (degrees)	20	10	30	uniform	(assumption)
Roof mass ($kg \cdot m^{-2}$)	2.64	2.26	3.58	uniform	Home Depot
Capital cost of existing plant (USD)	18,606,700	-	-	-	(Lubigi)
Electricity demand of existing plant ($kWh \cdot yr^{-1}$)	57,120	-	-	-	(Lubigi)
Staff of existing plant (people)	12	-	-	-	(Lubigi)
Salary for existing plant staff (million $UGX \cdot cap^{-1} \cdot month^{-1}$)	3.5	1	5	-	(Lubigi)
Capital cost of alternative plant (USD)	337,140	303,426	370,854	triangular	(CIDI)
Electricity demand of alternative plant ($kWh \cdot yr^{-1}$)	6,854	-	-	-	(assumption)
Skilled staff of alternative plant (people)	5	-	-	-	(CIDI)
Unskilled staff of alternative plant (people)	5	0	10	uniform	(CIDI)
Salary, skilled alt. plant staff (million $UGX \cdot cap^{-1} \cdot month^{-1}$)	5	1	5	-	(CIDI)
Salary, unskilled alt. plant staff (million $UGX \cdot cap^{-1} \cdot month^{-1}$)	0.75	0.50	1.00	uniform	(CIDI)
Electricity cost ($USD \cdot kWh^{-1}$)	0.17	0.08	0.21	triangular	(Umeme, 2019)
Electricity GHG impact factor ($kg\ CO_2eq \cdot kWh^{-1}$)	0.15	0.106	0.121	uniform	[329,336]
Existing plant lifetime (yr)	8	8	11	triangular	(Lubigi)
Alternative plant lifetime (yr)	50	45	55	Triangular	(CIDI)
Sewered population served by existing plant	40,000	30,000	50,000	uniform	(Lubigi)
Population producing latrine sludge treated by existing plant	416,667	375,000	458,333	triangular	Lubigi, CIDI, calculations
Population potentially served by alternative sludge treatment plant	50,000	45,000	55,000	triangular	(CIDI)
Greenhouse gas unit impact factors					
Concrete ($kg\ CO_2eq \cdot m^{-3}$)	300	218	385	uniform	[329]
Stainless steel ($kg\ CO_2eq \cdot kg^{-1}$)	4.33	3.07	5.5	uniform	[329]
Stainless steel sheet rolling ($kg\ CO_2eq \cdot kg^{-1}$)	0.65	0.58	0.71	uniform	[329]
Excavation ($kg\ CO_2eq \cdot m^{-3}$)	0.53	0.51	0.55	uniform	[329]
Plastic liner ($kg\ CO_2eq \cdot kg^{-1}$)	1.97	1.93	2.01	uniform	[329]
Gravel ($kg\ CO_2eq \cdot kg^{-1}$)	0.015	0.012	0.018	uniform	[329]

Table E.11. Parameter values, ranges, and distributions used in the reuse/disposal process stage of the quantitative modeling analysis.

Parameter	Expected	Low	High	Distribution	References
Crop application parameters					
Ammonia transfer losses (% of input ammonia)	5	0	10	uniform	[61]
N transfer losses (% of non-ammonia N)	2	0	5	uniform	(assumption)
P transfer losses (% of input)	2	0	5	uniform	(assumption)
K transfer losses (% of input)	2	0	5	uniform	(assumption)
Mg transfer losses (% of input)	2	0	5	uniform	(assumption)
Ca transfer losses (% of input)	2	0	5	uniform	(assumption)
C transfer losses (% of input)	2	0	5	uniform	(assumption)
N fertilizer price ($USD \cdot tonne\ N^{-1}$)	1,507	1,164	2,296	uniform	Kampala retailers (urea); [337]
P fertilizer price ($USD \cdot tonne\ P^{-1}$)	3,983	2,619	6,692	uniform	Kampala retailers (TSP); [337]
K fertilizer price ($USD \cdot tonne\ K^{-1}$)	1,333	1,214	1,474	uniform	Kampala retailers (MOP); [337]
Sludge fertilizer discount factor (nutrient price in sludge·price in commercial fertilizer^{-1})	0.1	0.04	0.13	uniform	Lubigi; model calculations
N fertilizer emissions (kg CO_2eq·kg N fertilizer produced^{-1})	5.4	1.8	8.9	triangular	[329]
P fertilizer emissions (kg CO_2eq·kg P fertilizer produced^{-1})	4.9	4.3	5.4	triangular	[329]
K fertilizer emissions (kg CO_2eq·kg K fertilizer produced^{-1})	1.5	1.1	2	triangular	[329]
Biogas fuel parameters					
Biogas transfer losses (%)	10	0	20	uniform	[151]
LPG selling price ($UGX \cdot kg^{-1}$)	6,500	6,077	6,667	uniform	(Kampala retailers)
LPG specific energy ($MJ \cdot kg^{-1}$)	50	49.5	50.4	uniform	[338]
LPG emissions (kg CO_2eq·kg LPG^{-1})	3	2.93	3.05	uniform	[341]

APPENDIX F: RESULTS FROM HOUSEHOLD SURVEYS IN BWAISE, UGANDA

A supplemental electronic file includes a spreadsheet showing summarized results from the household surveys conducted in Bwaise, Uganda (see Survey E.1 in Appendix E for the full survey instrument). The spreadsheet separates results into several tabs, based on topic area: Demographics and Assets; Water; Sanitation; Hygiene and Child Health; Energy; Agriculture; Livestock; Diet. The file name is as follows:

Trimmer_2019_Dissertation_-_Bwaise_survey_results_summary.xlsx

APPENDIX G: ETHICAL APPROVALS

Data collection activities associated with the application of the sanitation SES framework in Bwaise, Uganda (Chapter 6) underwent ethical review and approval by the Office for the Protection of Research Subjects at the University of Illinois at Urbana-Champaign, the Research Ethics Committee at Makerere University, and the Uganda National Council for Science and Technology. Approval letters from these three institutions are reproduced on the following pages.

UNIVERSITY OF ILLINOIS
AT URBANA-CHAMPAIGN

Office of the Vice Chancellor for Research
Office for the Protection of Research Subjects
805 West Pennsylvania Ave
Urbana, IL 61801

November 16, 2017

Jeremy Guest
Civil & Environmental Eng
3221 Newmark Civil Engineering Laboratory
205 N Mathews Ave
Urbana, IL 61801

RE: *Water and Sanitation Infrastructure Hygiene Behavior Fecal Environmental Contamination and Child Health in the Kiryandongo Refugee Settlement in Uganda*
IRB Protocol Number: 18225

Dear Dr. Guest:

This letter authorizes the use of human subjects in your project entitled *Water and Sanitation Infrastructure Hygiene Behavior Fecal Environmental Contamination and Child Health in the Kiryandongo Refugee Settlement in Uganda.* The University of Illinois at Urbana-Champaign Institutional Review Board (IRB) approved, by expedited review, the protocol as described in your IRB application. The expiration date for this protocol, IRB number 18225, is 11/12/2020. The risk designation applied to your project is *no more than minimal risk.*

Copies of the attached date-stamped consent form(s) must be used in obtaining informed consent. If there is a need to revise or alter the consent form(s), please submit the revised form(s) for IRB review, approval, and date-stamping prior to use.

Under applicable regulations, no changes to procedures involving human subjects may be made without prior IRB review and approval. The regulations also require that you promptly notify the IRB of any problems involving human subjects, including unanticipated side effects, adverse reactions, and any injuries or complications that arise during the project.

You were granted a three-year approval. If there are any changes to the protocol that result in your study becoming ineligible for the extended approval period, the RPI is responsible for immediately notifying the IRB via an amendment. The protocol will be issued a modified expiration date accordingly.

If you have any questions about the IRB process, or if you need assistance at any time, please feel free to contact me at the OPRS office, or visit our website at https://www.oprs.research.illinois.edu.

Sincerely,

Jennifer Ford

Jennifer Ford, MS
Human Subjects Research Specialist, Office for the Protection of Research Subjects

Attachment(s): 1 Research Team Application, 1 Waiver of Documentation, 8 Consent Forms

c: Assata Zerai

MAKERERE UNIVERSITY

P .O. Box 7062,
Kampala, Uganda
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Tel: 256-41-545040/0712 207926
Fax: 256-41-530185
E-mail: makssrec@gmail.com

COLLEGE OF HUMANITIES AND SOCIAL SCIENCES
SCHOOL OF SOCIAL SCIENCES
RESEARCH ETHICS COMMITTEE

Your Ref:

Our Ref: MAKSS REC 10.17.112

9th January 2018

Jeremy S. Guest, Ph.D.
Principal Investigator (MAKSS REC 12.17.112)
Assistant Professor
Center for Advanced Study Fellow
Excellence Faculty Fellow
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Phone: (217) 244-9247
E-Mail: jsguest@illinois.edu

Initial – Expedited Review
Re: Approval of Protocol titled: "Water and Sanitation Infrastructure hygiene behaviour, fecal environmental contamination, and child health in the Kiryandongo Refugee Settlement in Uganda"

This is to inform you that, the Makerere University School of Social Sciences Research Ethics Committee (MAKSS REC) granted approval to the above referenced study. The MAKSS REC reviewed the proposal using the Expedited review on **14th December 2017**. This has been done in line with the investigator's subsequent letter addressing comments and suggestions.

Your study protocol number with MAKSS REC is **MAKSS REC 12.17.112.** Please be sure to reference this number in any correspondence with MAKSS REC. Note that, the initial approval date for your proposal by **MAKSS REC was 14th December 2017**. This is an annual approval and therefore; approval expires on **13th December 2018. You should use stamped consent forms and study tools/instruments while executing your field activities at all times.** However, continued approval is conditional upon your compliance with the following requirements.

Continued Review
In order to continue on this study (including data analysis) beyond the expiration date, Makerere University School of Social Sciences (MAKSS REC) must re-approve the protocol after conducting a substantive meaningful, continuing review. This means that you must submit a continuing report Form as a request for continuing review. To avoid a lapse, you should submit the request six (6) to eight (8) weeks before the lapse date. Please use the forms supplied by our office.

1

Makerere University School of Social Sciences Research Ethics Committee
Valid Thru
13 DEC 2018

UNCST

Our Ref: HS 2424 **23rd July 2018**

Dear Assist. Prof. Guest,

Re: Research Approval: Water and Sanitation Infrastructure, Hygiene Behavior, Fecal Environmental Condition and Child Health in Resource – Limited Settings in Uganda

I am pleased to inform you that on **14/05/2018** , the Uganda National Council for Science and Technology (UNCST) approved the above referenced research project. The Approval of the research project is for the period of**14/05/2018 to 14/05/2020 .**

Your research registration number with the UNCST is **HS 2424.** Please, cite this number in all your future correspondences with UNCST in respect of the above research project.

As Principal Investigator of the research project, you are responsible for fulfilling the following requirements of approval:

1. All co-investigators must be kept informed of the status of the research.
2. Changes, amendments, and addenda to the research protocol or the consent form (where applicable) must be submitted to the designated Research Ethics Committee (REC) or Lead Agency for re-review and approval **prior** to the activation of the changes. UNCST must be notified of the approved changes within five working days.
3. For clinical trials, all serious adverse events must be reported promptly to the designated local IRC for review with copies to the National Drug Authority.
4. Unanticipated problems involving risks to research subjects/participants or other must be reported promptly to the UNCST. New information that becomes available which could change the risk/benefit ratio must be submitted promptly for UNCST review.
5. Only approved study procedures are to be implemented. The UNCST may conduct impromptu audits of all study records.
6. An annual progress report and approval letter of continuation from the REC must be submitted electronically to UNCST. Failure to do so may result in termination of the research project.

LOCATION/CORRESPONDENCE

Plot 6 Kimera Road, Ntinda
P. O. Box 6884
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COMMUNICATION

TEL: (256) 414 705500
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Uganda National Council for Science and Technology

(Established by Act of Parliament of the Republic of Uganda)

UNCST

Below is a list of documents approved with this application:

	Document Title	Language	Version	Version Date
1.	Research proposal	English	N/A	N/A
2.	Household survey	English	N/A	N/A
3.	Waiver of documentation of informed consent forms	English, Swahili and Luganda	N/A	N/A
4.	Focus group recruitments	English, Swahili and Luganda	N/A	N/A
5.	Focus group questions for informal settlements	English	N/A	

Yours sincerely,

Isaac Makhuwa
For: Executive Secretary
UGANDA NATIONAL COUNCIL FOR SCIENCE AND TECHNOLOGY

Copied to: Chair, Makerere University School of Social Sciences, Research Ethics Committee

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